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Regulatory Impact Analysis for the Final Criteria for Municipal Solid Waste Landfills

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This Regulatory Impact Analysis was prepared to evalue the U.S. EPA's revisions to Subtitle D criteria for municipal solid waste landfills. These regulations are a major rulemaking according to Executive Order 12291. Therefore, an RIA is required as part of this rulemaking. The analysis in this report evaluates the hybrid approach relative to four regulatory alternatives in terms of costs, economic impacts, impacts on small entities, health risk, and resource damage and discusses the overall rationale for the Agency's final choice. The RIA also includes a RFA that asesses impacts to small entities.

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December 1990

Note To Readers:

This Regulatory Impact Analysis was completed in 1990, prior to final revisions in the criteria.

Revised cost, economic impact and risk results for the Final Criteria for Municipal Solid Waste Landfills are presented in the:

> Addendum to the Regulatory Impact Analyses
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Regulatory Impact Analysis for the Final Criteria for Municipal Solid Waste Landfills

Prepared for

Regulatory Analysis Branch
Office of Solid Waste
U.S. Environmental Protection Agency

Prepared by

Temple, Barker & Sloane/Clayton Environmental Consultants
ICF, Incorporated
DPRA Incorporated
American Management Systems, Inc.

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Contents

Acknowledgments

Executive Summary

- Introduction
- Summary of the Final Rule
- Methodology
- Results
- Limitations

I. Introduction

- The Federal Role in Solid Waste Disposal
- The Nature of the Problem
- Required Analyses
- Outline of the Report

II. Background

- Current Regulatory Requirements
- Characteristics of Affected Landfills

III. Regulatory Options

- Final Rule
- Alternative 1: 1988 Proposal
- Alternative 2: Subtitle C Design Standards
- Alternative 3: Categorical Design Standards
- Alternative 4: Statutory Minimum

IV. Methodology: Cost Analysis

- Design and Cost Model
- Applying the Model to the Baseline and Regulatory Options
- Developing National Cost Estimates
- Cost Adjustments Based on Changes in Waste Management Approaches
- Summary

V. Methodology: Economic Impacts

  Overview of Approach
  Measures of Economic Impact
  Estimating Compliance Costs for Municipalities
  Database Description
  Municipal Financial Capability Baseline
  Waste Shifts and Potential Mitigating Factors

VI. Methodology: Risk and Resource Damage

  Introduction
  The Subtitle D Risk Model
  Model Inputs and Supporting Analyses
  Environmental Settings
  Leachate Characterization
  Impacts
  Exposure Distance and Populations

VII. Results: Cost Analysis

  Baseline
  Final Rule
  Alternative 1: 1988 Proposal
  Alternative 2: Subtitle C Design Standards
  Alternative 3: Categorical Design Standards
  Alternative 4: Statutory Minimum
  Cost Summary
  Adjustments to the Costs for Regionalization and for Changes in Future Waste Discards
  Conclusion

VIII. Results: Economic Impacts

  Impact Analysis Results
  Comparison of Impacts and Financial Capability
  Regulatory Flexibility Analysis

IX. Results: Risk and Resource Damage

  Health Risk
  Resource Damage

X. Comparison of Costs and Benefits

  Costs and Risk Reduction
  Costs and Resource Damage Reduction

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This report and its supporting analyses have been under development for more than five years, involving a team of contractors from Temple, Barker & Sloane, Inc./Clayton Environmental Consultants; ICF, Inc.; Pope-Reid Associates, Inc. (now DPRA); and American Management Systems, Inc. The project was under the direction of Ron Burke, the EPA Project Officer from the Office of Solid Waste from 1985 to 1990. The EPA Project Officer since 1990 has been Sara Rasmussen.

The EPA Project Officer would like to acknowledge the assistance of several individuals who contributed significantly to the development of this study. Cliff Rothenstein provided guidance, support, and review of the draft documents in the early stages of the analysis. Debbie Dobkowski has also provided guidance and support. Several other individuals within EPA also assisted in developing this document. In particular, Mike Flynn of the Municipal Waste Program provided detailed direction on the content of the regulatory requirements. Donna Perla directed the development of the Risk Model. Dave Croxton reviewed and commented on draft documents, and Jim Craig provided guidance on the use of the facility survey results.

The EPA Project Officer also wishes to thank TBS/Clayton, ICF, DPRA, and AMS for their many long hours spent developing this document, often under tight deadlines. In particular, the substantial support provided by several individuals should be recognized.

At TBS/Clayton, special thanks go to Deems Buell, Kevin Dietly, David Riordan, and Patty Robertson. TBS/Clayton performed the economic impact analysis, assisted DPRA in the cost analysis, and characterized the MSWLF baseline. TBS/Clayton also assimilated the various analyses into this document.

At ICF, the efforts of Randy Freed, Joanne Colt, Tom Gherlein, Elaine Hammick, John Trever, and Gaynor Dawson are acknowledged. ICF performed the benefits analysis, modified the Liner Location Model, developed simulation models for leachate quality, and determined the environmental settings of MSWLFs.

At DPRA, the efforts of Chris Lough, William Murphy-Rohrer, Roger Laitinen, Shirley Smith, Mary Sands, and Brian Ross are appreciated. DPRA developed the cost model and modified the failure/release component of the Liner Location Model.

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Executive Summary

Introduction

This Regulatory Impact Analysis (RIA) was prepared to evaluate the U.S. Environmental Protection Agency's (EPA's) revisions to Subtitle D criteria for municipal solid waste landfills (MSWLFs). These regulations are a major rule-making according to Executive Order 12291 (aggregate compliance costs are greater than $100 million per year); therefore, an RIA is required as part of this rule-making. The analysis in this report evaluates the hybrid approach (referred to in this RIA as the final rule) relative to four regulatory alternatives in terms of costs, economic impacts, impacts on small entities, health risk, and resource damage and discusses the overall rationale for the Agency's final choice. The RIA also includes a Regulatory Flexibility Analysis (RFA) that assesses impacts to small entities.

Highlights of the Analysis

In addition to the final rule, the Agency looked at four regulatory options in this analysis. These regulatory options include three additional preventive approaches (approaches that prevent contamination of ground water from releases): a Subtitle C approach, the 1988 proposal, and a categorical approach. One pollute-and-clean-up approach was also considered: the statutory minimum. A summary of these options is contained in Table 1.

This analysis represents EPA's best efforts to quantify the costs, economic impacts, and benefits (health risk and resource damage) of the regulatory options. It should be noted, however, that as in any analysis, the results are necessarily based upon incomplete data and on simplified assumptions. A discussion of limitations of the analyses is included in the final section of this Executive Summary.

Table 1

Regulatory Options Considered in Analyses

Preventive Oriented	
Final Rule	■ Mix of uniform design, MCL performance standard ■ Good management practices ■ Ground-water monitoring ■ Corrective action ■ Closure/post-closure care ■ Location standards
1988 Proposal (Alternative 1)	■ Risk-based performance standard ■ Good management practices ■ Ground-water monitoring ■ Corrective action ■ Closure/post-closure care ■ Location standards
Subtitle C (Alternative 2)	■ Double liner, composite cover ■ Good management practices ■ Ground-water monitoring ■ Corrective action ■ Closure/post-closure care ■ Location standards
Categorical (Alternative 3)	■ Performance design criteria based upon categories of environmental settings ■ Good management practices ■ Ground-water monitoring ■ Corrective action ■ Closure/post-closure care ■ Location standards
Reactive Oriented	
Statutory Minimum (Alternative 4)	■ Ground-water monitoring ■ Corrective action ■ Location standards

The analyses of cost, economic impact, and small community impact were based upon the calculation of incremental costs – costs associated with compliance of the regulations. These costs are presented in a range, with the lower end (10-year post-closure care period, shifts to regionalization, and increased waste diversion) and upper ends (40-year post-closure care period, no regionalization or increased waste diversion) referred to as the lower and upper bounds, respectively. However, EPA feels these estimates represent a reasonable range of results and not absolute bounds. The combined annualized costs (for new and existing facilities) of the five options range from $447 million for the statutory minimum with the lower-bound scenario, up to $2,844 million for the Subtitle C approach under the upper-bound scenario. The final rule falls near the lower end of this range, at $620 million assuming the lower-bound scenario and $1,038 million for the upper-bound.

Two economic impact measures are cost as a percentage of community expenditure (CPE) and cost per household (CPH). The upper-bound cost estimates (the most conservative) are used to compute these impact measures. The final rule exceeds 1 percent of CPE for 27 percent of the communities. The 1988 proposal has similar impacts. For this measure, the categorical approach and the Subtitle C approach have broader impacts than the final rule, exceeding the 1 percent threshold in 36 and 64 percent of communities, respectively. The statutory minimum exceeds this threshold for 14 percent of the communities. For the second measure, CPH, the final rule exceeds the moderate impact threshold of $100 per household per year in fewer than 0.1 percent of the communities. The Subtitle C approach and statutory minimum (due to high clean-up costs at some sites) exceed this moderate impact threshold for 20 percent and 6 percent of the communities, respectively.

The benefits analysis evaluated risk and resource damage due to contamination of ground water used for drinking water. The analysis estimates reduction in cancer cases, population risk, and resource damage (valued as the cost to replace contaminated ground water). The regulatory options all achieve roughly similar results for reduction in human health risks from drinking contaminated ground water. The final rule and the statutory minimum eliminate 70 percent (16 cases) of the predicted cancer cases caused from one set of new landfills with a 20-year lifespan. The Subtitle C approach and the categorical approach are more effective (eliminating 19 and 18 cases, respectively); the 1988 proposal is less effective, eliminating 15 cases. However, with regard to resource damage, the preventive approaches are more effective. Baseline resource damage for one set of new landfills with 20-year lifespans is estimated to be $2.6 billion. The Subtitle C and the categorical approaches are most effective in reducing resource damage (by $1.3 billion to $1.4 billion and by $1.7 billion to $2.1 billion, respectively). The final rule falls in the middle of the range, reducing resource damage by $1.3 billion to $1.7 billion. The 1988 proposal and the statutory minimum are least effective, reducing resource damage by $1.3 billion to $1.4 billion and by $0.9 billion to $1.0 billion, respectively.

The analysis has omitted several hard-to-quantify benefits of the final rule's prevention approach versus a pollute-and-clean-up approach. These benefits include protecting the existence value of ground water, greater certainty associated with the prevention approach versus reliance on ground-water monitoring and clean-up technologies, reduced potential for intergenerational inequities, reduced potential that landfills will become a federal burden, reduced social welfare impacts (i.e., limiting impacts on property values), avoidance of an extremely costly future burden on financially ill-prepared communities for corrective action, internalization of the cost of environmental impact, and increased public confidence in landfills.[1] Because of these factors, the quantitative portion of this RIA should be used as only one tool, in conjunction with an assessment of nonquantified factors, when evaluating the various regulatory alternatives presented here.

Characteristics of the Municipal Solid Waste Universe

The characteristics of MSWLFs, as reported in the EPA municipal landfill survey, vary significantly across the country on a number of parameters.

Number of Landfills and Waste Volumes

On the basis of the EPA Solid Waste (Municipal) Survey, EPA estimates that there were 6,034 active municipal solid waste landfills in the United States in 1986. The survey indicated that these landfills accepted a total of approximately 208 million tons per year (approximately 800,000 tons per day [TPD]).

Lifespan

The landfill survey results show that 55 percent of the MSWLFs in operation at the time of the survey planned to close by 1997; another 25 percent planned to close in the following 10 years. In other words, 80 percent of all municipal solid waste landfills in operation at the time of the survey will have closed by 2007 and will have to be replaced with new landfill capacity or other disposal capacity. Of the remaining 20 percent of landfills, which will operate beyond 2007, 5 percent are projected to continue operation for 100 years or more.

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[1] Further explanation of these benefits is provided in the public docket for this rule.

Size

The reported sizes of landfills cover a wide range from very small local landfills receiving one ton or less of waste each day to large landfills that receive thousands of tons of waste per day. Most landfills are small, with 51 percent of all landfills estimated to be receiving less than 17.5 TPD of waste and 81 percent receiving less than 125 TPD. In terms of total waste disposed, however, the large facilities receive a disproportionate amount of all municipal solid waste (MSW) generated in the United States (landfills receiving more than 1,125 TPD handle 40 percent of all waste generated but account for only 3 percent of all MSWLFs).

Ownership

Most MSWLFs (81 percent) are owned by local governments, including counties, cities, towns, and villages. Approximately 15 percent of all landfills are privately owned. The remaining 4 percent of landfills are owned by the federal government at national parks, military bases, and other federal installations and by state governments at parks, universities, and other locations.

Trends

The amount of solid waste generation per person has been increasing (48 percent between 1960 and 1988[2]) and is expected to continue to do so, while the number of landfills has been declining and is expected to decrease in the coming years. As current landfills close, communities are faced with the increasingly difficult task of siting a landfill. Siting landfills has become difficult for several reasons – political factors and the NIMBY (not in my back yard) factor, for example. The increase in solid waste generation and the difficulty in siting new landfills have resulted in solid waste crises for several regions of the country. To solve their problems, cities are transporting waste great distances, sometimes across state boundaries, to large regional landfills. EPA expects this trend to continue in the future. In addition, communities are expected to consider supplemental efforts to reduce waste volume, through source separation and other programs, thereby potentially extending the life of existing landfills.

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[2] *Characterization of Municipal Solid Waste in the United States: 1990 Update*, EPA Office of Solid Waste, June 13, 1990, p. ES-9.

Characteristics That Affect Cost/Benefit Analyses

As noted, the characteristics of MSWLFs vary significantly on a number of parameters, many of which affect the cost/benefit analyses. These factors include location, size, ownership, and remaining life.

Location plays a strong role in both the risk and cost analyses. Environmental setting, including hydrogeology and net infiltration, affects the contaminants' fate, transport, and potential for ground-water contamination, thus affecting potential exposure and subsequent design criteria needed to reduce releases. Population density of the site is also important. Since results from the 1988 survey indicated that 54 percent of the landfills currently have no drinking-water wells within a one-mile radius, EPA assumes these landfills have no risk.

Landfill size affects cost because of the tremendous economies of scale in landfill operations (larger facilities can dispose of waste at a much lower unit cost).

The ownership variable affects economic impact because landfill owners bear the responsibility of complying with revised criteria. However, the ultimate cost of compliance will fall on landfill users, through increased user fees, higher taxes, or a reallocation of funds within the local municipality's budget (resulting in reduced services elsewhere).

Remaining life is important because it establishes the remaining time period during which existing facilities can recover the cost of complying with this regulation. If costs cannot be recovered during a short remaining life, other funding sources – such as a surcharge on ongoing disposal operations or a transfer of tax revenues – will be required. Remaining life is also important to cost/benefit analysis because existing and new landfills will be regulated differently.

Summary of the Final Rule

EPA chose a prevention-based approach for the final rule (referred to as the hybrid approach in the Federal Register). EPA believes prevention is a better strategy than clean-up of pollution for maximizing the hard-to-quantify environmental benefits cited earlier in this summary.

The final rule includes design criteria (described below), location standards, operating and maintenance standards (including disease vector control and gas monitoring), closure and post-closure care, ground-water monitoring, corrective action, and financial assurance. Most of these requirements are applied uniformly to all MSWLFs. The rule provides flexibility to states with federally approved Subtitle D programs in choosing alternate criteria for post-closure care periods, small rural community exemptions, and corrective action triggers.

With regard to design criteria, the final rule includes flexibility by providing both a uniform design and, for states with an approved program, the option of using performance-based design criteria. This dual approach recognizes three differences within the landfill universe. First, landfill characteristics vary widely and not all locations require the same controls to achieve environmental protection. Second, some states will have approved Subtitle D programs, while others will not. Third, existing landfills will face different constraints than new landfills in meeting standards.

Design Criteria for Existing Landfills

Upon closure, all existing landfills will be required to protect human health and environment by installing a final cover designed to minimize the formation of leachate. Retrofitting liners and leachate collection systems (LCSs) onto existing units was rejected by EPA as impractical, possibly environmentally hazardous, and disruptive of ongoing solid waste management activity.

Design Criteria for New Landfills in Approved States

New landfills sited in approved states will have the choice either to use the uniform design standard (discussed below) or design the landfill based upon a performance standard. The MCL-based performance standard (based on maximum contaminant levels defined in 40 CFR 264) requires that no releases from the unit exceed any MCLs (or background levels if no MCL exists) at the unit boundary or alternate boundary as determined by an approved state for any constituents during the active life of the landfill plus the post-closure care period. Approved states have the option of using health-based criteria for constituents with no MCLs if they are protective of human health.

Design Criteria for New Landfills in Non-Approved States

A uniform design standard will apply to new landfills in non-approved states: Units must have a composite liner consisting of a flexible membrane liner (FML) and at least two feet of compacted soil with a hydraulic conductivity of no more than 1×10^{-7} centimeters per second. A leachate collection system is also required. In addition, the cover material must be no more permeable than the bottom liner.

Ground-Water Monitoring

New landfills must have ground-water monitoring (GWM) in place prior to operation. GWM requirements for existing landfills are phased in over five years. All landfills are required to continue GWM throughout the post-closure care period.

Closure/Post-Closure

All landfills are required to maintain the cover and continue ground-water monitoring during the duration of the post-closure care period. The rule requires a 30-year post-closure care period, but allows approved states the flexibility to chose an alternate period.

General Facility Requirements

General facility requirements in the final rule include run-on/run-off controls, exclusion of hazardous waste, standards for daily cover, on-site disease vector control, routine methane monitoring, liquids restrictions, elimination of open burning, public access control, recordkeeping and financial assurance.

Location Standards

The final rule includes location restrictions regarding airports, floodplains, wetlands, fault areas, seismic impact zones, and unstable areas.

Corrective Action

Corrective action will be triggered for landfills upon detection at the unit boundary (or alternate boundary, if specified in an approved state) of a statistically significant increase of Appendix II constituents above ground-water protection standards.

Small Rural Community Exemption

An exemption from design, ground-water monitoring, and corrective action requirements may be provided by approved states to small landfills (less than 10 tons per day) serving small communities (fewer than 5,000 residents). This exemption will be allowed only if no evidence exists of ground-water contamination from the landfill, and either the landfill is located in a geographically isolated community (a community experiencing an annual interruption of at least three consecutive months of surface transportation) or there is no practicable waste management alternative and the area receives 25 inches of precipitation or less.

Methodology

Modeling Assumptions for Final Rule and Regulatory Alternatives

EPA analyzed the final rule and four regulatory alternatives in the RIA. These alternatives allow for a cost and benefit comparison, as required under Executive Order 12291. See Table 1 for a summary of the options.

To reflect the flexibility in the rule regarding post-closure care periods, the costs and effectiveness of all regulatory options were modeled with post-closure care periods of 10 years and 40 years (except the 1988 proposal, which was modeled with post-closure care periods of 30 and 40 years).

Financial assurance, most location standards, and the small rural community exemption were not incorporated into the analysis.

Final Rule (Hybrid Approach)

To estimate the costs and benefits of the final rule, we developed an approach to simulate how municipalities would respond to the flexibility provided in the rule.

New Units – Uniform Design. We assigned the uniform design (the most stringent) to new units in both approved and non-approved states, located in environmental settings that are likely to pose the greatest potential for health risk and resource damage (such as wet climates with short travel times for the migration of constituents from the landfill to the water table). We assigned the uniform design to 27 percent of new landfills; the assumptions are discussed further in Chapter IV.

New Units – Performance Standards. We assumed that the remaining 73 percent of landfills will have flexibility in meeting the design criteria by complying with the health-based (MCL) performance standard rather than uniform design standards. To simulate compliance with the performance standard, we assigned units to one of three designs: the baseline design, which assumes an unlined landfill with a cover consisting of earth and vegetation; an unlined design with a synthetic cover system; and a landfill with a synthetic cover system, a synthetic liner, and a leachate collection system. These three designs were assigned to landfills for modeling purposes only; facilities could use other designs to comply with the performance standard.

We used results from the Subtitle D Risk Model (discussed later) to indicate which landfills would meet the performance standards under baseline conditions. These landfills were assigned an unlined design with a vegetative cover. For landfills where concentrations of constituents would exceed the performance standards, more elaborate designs (including synthetic covers, synthetic liners, and leachate collection systems) were assigned to minimize or eliminate releases. In fact, facilities in approved states could also be designed to the uniform standard.

Table 2 summarizes the assignment of landfills to designs for modeling purposes in the RIA. Most of the landfills (56 percent) are assigned the unlined/vegetative cover design.

Table 2

Summary of Design Assignments for the Final Rule
New Landfills Only

	Percentage of Landfills
Performance Standard	
Unlined/vegetative cover	56%
Unlined/synthetic cover	<1
Synthetic liner/synthetic cover	16
Uniform Standard	
Composite liner/synthetic cover	27
Total	100%

Existing Landfills. To simulate the impact of the final rule standards (minimize the formation of leachate to protect human health and the environment), we assigned either a synthetic cover or a vegetative final cover to landfills based on their potential for generating leachate.

Additional Provisions Modeled. Other provisions modeled include post-closure care, gas monitoring, run-on and run-off controls, ground-water monitoring, and corrective action.

Alternative 1 – 1988 proposal

Alternative 1 reflects the requirements of the 1988 proposed revisions to the criteria. This alternative includes risk-based performance standards that landfills must meet. These standards differ for new and existing landfill units as discussed below. In addition, Alternative 1 includes a set of general facility standards similar to those of the final rule.

New Units. For new units, the performance standard for design is linked to a health- and environment-based ground-water risk level called the design goal. The proposal specifies only that the design goal for ground water must fall in the protective range of risk levels between 10^{-4} and 10^{-7}; individual states are responsible for choosing the relevant design goal for units in their states. The proposal also does not set specific designs that landfill owners must use to meet the design goal. Any controls approved by the state are acceptable if they will result in the landfill meeting the performance standard.

To model Alternative 1, we selected one design goal of 10^{-5} and modeled the same three designs modeled for the performance standard under the final rule. To assign designs, we compared the design goal with constituent concentrations that would result in a 300-year, average, "most exposed individual" (MEI) risk to humans of 10^{-5}. The use of the average MEI risk over 300 years was an assumption made for this analysis only.

Table 3 summarizes the landfill designs that we assumed would meet the performance standard. As in our modeling of the final rule, we assigned designs to actual landfills based on their estimated releases. The majority of landfills are not assigned additional controls beyond the general facility standards (i.e., they can meet the performance standard under baseline conditions).

Table 3

Summary of Design Assignments for Alternative 1
New Landfills Only

	Percentage of Units
Design	
Unlined/vegetative cover	61%
Unlined/synthetic cover	11
Synthetic liner/synthetic cover	28

Existing Units. The performance standard requires that the final cover prevent infiltration into the waste. To simulate this standard, we used the same assignments of vegetative and synthetic covers to landfills as we used for the final rule.

Other Requirements. The general facility standards are similar to those for the final rule. These include post-closure care (which is analyzed at durations of 30 and 40 years), ground-water monitoring (which includes more sampling parameters than the final rule requirements), gas monitoring, run-on and run-off controls, and corrective action. Unlike the final rule and the other three alternatives, the trigger levels for corrective action are based on a 10^{-5} risk level.

Alternative 2 – Subtitle C Approach

Alternative 2, a Subtitle C type design, is more stringent than the final rule. To simulate this option, we modeled new landfills with a redundant containment system, similar to that specified for hazardous waste landfills under Subtitle C of the Resource Conservation and Recovery Act (RCRA). The system includes two synthetic layers, with leachate collection systems above each synthetic, and a supporting clay layer to delay leachate migration if leachate escapes the two synthetic layers. We modeled a composite cover system at closure for both new and existing units.

Other Requirements. We modeled more elaborate ground-water monitoring sampling and testing procedures than under the final rule. All the general facility requirements in the final rule are imposed on landfills under this option. These include corrective action, gas monitoring, run-on and run-off controls, and post-closure care.

Alternative 3 – Categorical Approach

Alternative 3 categorizes landfills according to their potential to generate leachate and assigns designs to prevent releases from migrating into the aquifer. The two major criteria used to assign designs are that the cover must minimize infiltration into the waste after closure, and that containment and leachate collection systems must prevent releases to the underlying aquifer during the active life of the landfill. Using these criteria and several assumptions described in Chapter IV of the RIA, we assigned landfills to one of seven designs, presented in Table 4. These designs are not specifically required under this alternative, but for modeling purposes we simulated landfill performance and developed designs that we estimated would meet the criteria. The most common design assigned is also the most expensive: The synthetic liner/synthetic cover with a leachate collection system was assigned to 56 percent of the landfills. Also, for this alternative we modeled the same general facility standards for landfills as modeled for the final rule.

Table 4

Summary of Design Assignments for Alternative 3
New Landfills Only

LCS	Containment	Cover	Percentage of MSWLFs
No	Unlined	Vegetative	18%
No	Unlined	Synthetic	2
No	Clay	Vegetative	5
Yes	Synthetic	Vegetative	8
Yes	Synthetic	Synthetic	56
Yes	Unlined	Vegetative	4
Yes	Unlined	Synthetic	8

Alternative 4 – Statutory Minimum

Alternative 4, the statutory minimum, is the least stringent option: It imposes no design standards on landfills and no general facility standards. The only requirements modeled are ground-water monitoring during the active life and a post-closure care period and corrective action as necessary. The emphasis of this option is on response to releases instead of prevention. This alternative is similar to the statutory minimum described in the 1984 Hazardous and Solid Waste Amendments (HSWA) to RCRA, except that no location standards are included.

Analytical Methodology

The RIA addresses three major aspects of the regulatory options: cost, economic impact, and risk and resource damage. Our methodology for analyzing each of these components is described briefly below and in greater detail in Chapters IV, V, and VI.

Cost Methodology

All facilities must comply with the current Subtitle D Criteria (40 CFR Part 257) promulgated by EPA in 1979. While EPA established these criteria, each state has the responsibility of ensuring that its MSWLFs meet these criteria as well as any other requirements that the state may have. This analysis looks at the baseline costs of landfills, including current state requirements, and estimates the incremental costs to facilities due to the regulatory options.

These cost estimates assume that the current volume of waste landfilled remains constant and that, as landfills close, they are replaced with new units of the same size. In reality, however, the solid waste management system is changing. We incorporated two sensitivity analyses that reflect these changes into the cost analysis: regionalization (shifting waste from small landfills to larger, regional facilities) and shifts away from disposal resulting from source reduction, recycling, and combustion.

Subtitle D Cost Model

The basis of the cost analysis is a facility design and cost model that takes selected input variables and produces the specifications and costs for a municipal solid waste landfill. This design and cost model first identifies the various steps involved in designing, constructing, operating, and closing a landfill; combines this information with data on costs and fees for the components; and estimates the total cost of the landfill. The cost components include a wide range of items such as land clearing, excavation, equipment, labor, liner materials, and ground-water monitoring wells.

We estimated the cost of constructing and operating a "baseline" landfill – one designed in accordance with current criteria and best engineering practices, but excluding many preventive features adopted in this rule-making. The national baseline disposal cost estimated by the model (annualized) was $2.1 billion. We also estimated the costs of building and operating landfills under each of the regulatory options. The incremental costs between the regulated units and the baseline units form the basis for the cost and impact results.

Accounting for Current State Regulations

We made several adjustments to incremental costs to take account of states with programs that already require ground-water monitoring or cover and containment systems.

- EPA identified 24 states (2,702 landfills) with equivalent GWM programs. Thus, for the landfills in those states, we excluded capital costs for ground-water studies and ground-water monitoring wells under each option.
- EPA concluded that 18 states (3,203 landfills) have containment system requirements that would be acceptable under the performance standards in the final rule and Alternative 1. In those states, landfills that were assigned a synthetic liner and LCS did not incur the incremental costs.
- EPA concluded that four states (454 landfills) have containment system requirements that are equivalent to the uniform standard under the final rule and cover requirements that are at least as stringent as those that may be required under the final rule's performance standard. Landfills in these

states incur no additional costs from federal containment or cover requirements under the final rule, the 1988 proposal, and Alternative 3.

- Two of the states also receive full credit for design requirements under Alternative 2.

The dollar value of these credits for the final rule was computed at $454 million (annualized) for a total baseline cost of $2.6 billion per year.

Assignment of Compliance Costs Across Landfills

Using the cost model, we estimated compliance costs for approximately 10 percent of all MSWLFs, based on responses to the MSWLF facility survey. Cost estimates were made for existing landfills and for replacement landfills meeting the specifications of the final rule and each alternative. A combined cost estimate, which averaged together costs at the existing facility plus the new replacement facility, was then made for each landfill. The combined cost estimates were scaled up to the universe of all MSWLFs to estimate national compliance costs.

Regionalization Analysis

The analysis of regionalization included a series of decision rules that govern the decision of a landfill owner to replace a closing landfill or to investigate sharing in a larger landfill. We identified combinations of landfill sizes and population densities where shifts to larger landfills would cost less than replacing the existing unit, based on disposal and transportation costs. We applied these rules to landfills as they closed in future years, based upon survey data, and estimated a new size distribution of landfills 20 years from the enactment date. The regionalization analysis (described in Chapter IV) reveals that, 20 years after the effective date of the regulation, up to 21 percent of the 800,937 tons per day landfilled will shift to larger units. All shifts occur from landfills that accept less than 275 tons per day.

Waste Diversion Analysis

We also assumed that shifts of waste away from landfill disposal (source reduction, recycling, and combustion) would reduce the cost of compliance with the rule and the alternatives. Because these shifts are already occurring, and are not resulting from the rule, we assumed that increased use of these options will simply reduce the amount of waste subject to incremental compliance costs. Based on EPA projections of disposal trends, we estimated an 18 percent decrease in landfilled waste volume 20 years after the effective date of the rule. We assume that this decrease will translate into a proportional decrease in total compliance costs.

Economic Impact Methodology

The economic impact analysis assessed the impacts of the regulatory options on those who would face incremental costs of the rule and analyzed how those impacts would be distributed. To provide the most conservative impact estimates, the analysis was conducted using the upper-bound cost scenario.

The analysis looks at impacts to municipalities and households. While approximately 15 percent of all MSWLFs are owned by private operators, the RIA did not assess impacts to private entities in detail. Identifying ownership and financial capability of owners of these landfills was outside the scope of the analyses. Further, MSW handled by these landfills comes from the governments or households that contract for the landfills' services. Thus, even in the case of private ownership, the governments and households served will ultimately bear the cost and impacts of new regulations.

Impact to Municipalities

EPA assessed the impacts to municipalities because 81 percent of all landfills that handle MSW are owned by local governments. Municipalities that own these landfills will bear the initial economic impacts of revised Subtitle D regulations.

To assess economic impact to municipalities, the analysis looked at cost as a percentage of expenditure (CPE) for the local governments, and assessed the financial strength of municipalities.

Impacts to Households

EPA also considered impacts on households since governments are likely to pass increased costs on to their own citizens (and to neighboring governments that might also use the landfill) in the form of increased taxes or fees, or decreased services of other types if the community is operating under tight budget constraints. Ultimately, the households that use the landfill to dispose of their waste will pay the increased costs of landfill operations. We estimated cost per household (CPH) and cost as a percentage of median household income (CPMHI) on a community-by-community basis.

Risk and Resource Damage Methodology

Scope

The benefits analysis evaluated risk and resource damage from contaminated ground water only. The analyses assess the adverse effects of releases from MSWLFs on human health and resource value, and examine the effectiveness of the regulatory options in reducing these impacts. Current state controls are not incorporated into the baseline for these analyses.

For the benefits analysis, we assumed that all facilities are new (modeling existing units was beyond the scope of this analysis). We assumed that the operating lifespan of the landfills was 20 years. We assumed that new landfills are constructed at sites with the same characteristics as the ones they replace, and are designed and operated throughout their lifetime according to the requirements of the option being considered. Unlike the cost analysis, which assumes perpetual replacement of landfills, the risk analysis does not address replacement landfills. All regulatory scenarios were analyzed with a 30-year post-closure care period for the human health risk modeling, and 10-year to 40-year post-closure care period for the resource damage analysis.

Subtitle D Risk Model

The model builds directly on the Liner Location Risk and Cost Analysis Model, developed to address these same questions for land disposal units regulated under Subtitle C of RCRA. It is a dynamic model that simulates environmental fate and transport and dose-response as deterministic processes, while simulating containment system failures stochastically. The model focuses on ground water as the environmental medium of concern; it does not evaluate the effects of contamination to surface water and the ecosystem.

Our basic approach was to use the Subtitle D Risk Model to simulate risk and resource damage first in the baseline, and then under each of the regulatory options being considered. We modeled risk and resource damage in hundreds of scenarios that represent unique combinations of landfill size, climatic and hydrogeologic setting, and exposure distance. Using preliminary results from EPA's Municipal Landfill Survey and a separate mapping effort, we then estimated the frequency with which each scenario occurs in the nationwide population of municipal landfills and weighted the results for each scenario to reflect its frequency of occurrence.

We modeled eight constituents of concern, which were selected based on their observed median concentrations in MSWLF leachate (taken from a database of leachate from 44 landfills), their toxicity, and other factors.

Risks

We report risks as average maximum exposed individual (MEI) risk (i.e., the actual most exposed individual's lifetime risk, based upon exposure at the estimated nearest well, averaged over the 300-year modeling period). Population risks are estimated by calculating size of plume, number of wells affected, and average population density served by wells as reported by the survey.

Resource Damage

Resource damage is measured as the present value of the cost to provide water (from a nearby ground-water source) to users whose supply is contaminated by releases from landfills. Damages are greatest in locations where there is current use of the ground water as drinking water. Where ground water is not currently in use, the probability of future use is incorporated into the analysis. A water supply is considered contaminated when constituents exceed the lower of regulatory limits under the Safe Drinking Water Act or taste and odor thresholds.

Results

Cost Results

In order to reflect uncertainty associated with the flexibility of the rule regarding post-closure care period as well as the uncertainty regarding size of replacement landfills and potential reduction in waste volume due to waste diversion in the future, cost results are presented for three scenarios:

- *Lower-bound* – 10-year post-closure care period and regionalization and waste diversion 20 years after the effective date
- A 10-year post-closure *care period*; no regionalization or waste diversion
- *Upper-bound* – A 40-year post-closure care period, with no regionalization or waste diversion

While EPA refers to the two ends of the ranges presented as the upper- and lower-bounds, EPA believes that these represent a reasonable range of results other than absolute bounds.

National Annualized Incremental Costs

The combined annualized costs (for new and existing facilities) of the five options range from $450 million, for Alternative 4 with the lower-bound scenario, up to $2,840 million for Alternative 2, with the upper-bound scenario. The final rule falls near the lower end of this range, at $620 million assuming the lower-bound scenario and $1,040 million for the upper-bound. The costs under all three cost scenarios are summarized in Table 5. The order of the options from least expensive to most expensive remains the same regardless of the cost scenario: The final rule always is in the middle of the five options.

The lower bound estimates compliance costs 20 years after the effective date. It includes estimated shifts to larger regionalized facilities resulting in annual savings of up to $219 million for the final rule and savings of $136 million resulting from waste diversion (recycling, shift to combustion, etc.).

Compared to modeled baseline costs, the final rule represents an increase of between 24 and 40 percent. The baseline costs including state requirements were computed by the cost model at $2.6 billion. The final rule adds between $620 million and $1,040 million to the total baseline cost.

Table 5

Summary of Annualized Compliance Costs for All Options
(dollars in millions)

	Lower Bound[1]	10-Year Post-Closure Care Period[2]	Upper Bound[3]
Final Rule	$620	$980	$1,040
Alternative 1 (Proposal)	600	940	970
Alternative 2 (Subtitle C Design)	1,750	2,750	2,840
Alternative 3 (Categorical Approach)	740	1,170	1,230
Alternative 4 (Statutory Minimum)	450	700	760

[1] Lower bound assumes a 10-year post-closure care period for all options except Alternative 1, where post-closure care lasts 30 years; costs include adjustment for regionalization and diversion of waste to recycling and combustion.

[2] Ten-year post-closure care except for a 30-year period for Alternative 1; no adjustment for waste diversion or regionalization.

[3] Upper bound assumes a 40-year post-closure care period for all options.

Corrective Action

Corrective action, determined through modeling runs, is triggered under all options for a portion of existing landfills. Corrective action is triggered only at new landfills under Alternative 4 (statutory minimum).

Corrective action costs contribute significantly to the cost of Alternative 4, as shown in Table 6. This option has the least-stringent design standards; therefore, more corrective actions are necessary. Alternative 1 has the next largest percentage of costs from corrective action, but the percentage is dramatically lower than for the statutory minimum. Note that we did not compute the corrective action cost percentage for the lower-bound estimate – only for the shorter post-closure scenario and the upper bound.

Table 6

Percentage Contribution of Corrective Action to Annualized Compliance Costs for All Options[1]

(dollars in millions)

	10-Year Post-Closure Care Period[2]	
	Total Cost Including Corrective Action[3]	**Percentage from Corrective Action**
Final Rule	$980	5.0%
Alternative 1 (Proposal)	940	5.2
Alternative 2 (Subtitle C Design)	2,750	1.8
Alternative 3 (Categorical Approach)	1,170	4.2
Alternative 4 (Statutory Minimum)	700	74.7

	Upper Bound[3]	
	Total Cost Including Corrective Action	**Percentage from Corrective Action**
Final Rule	$1,040	4.7%
Alternative 1 (Proposal)	970	5.0
Alternative 2 (Subtitle C Design)	2,840	1.7
Alternative 3 (Categorical Approach)	1,230	4.0
Alternative 4 (Statutory Minimum)	760	69.5

[1] Corrective action for the final rule and Alternatives 1, 2, and 3 are for corrective action at existing landfills only. No new landfills trigger corrective actions under these options.

[2] Ten-year post-closure care period for all options except Alternative 1, which has 30 years; no adjustment for additional waste diversion or regionalization.

[3] Forty-year post-closure care period for all options; no adjustment for additional waste diversion or regionalization.

Distribution of Cost

The distribution of costs for specific components of the rule varies between the preventive approaches and the statutory minimum. Table 7 presents the major cost components for the final rule and the statutory minimum. Corrective action accounts for close to 70 percent of the costs under the statutory minimum, while it accounts for much less (5 percent) with the final rule.

Table 7

Breakout of Total Annualized Costs

	Final Rule Upper Bound	Statutory Minimum Upper Bound
Liners, covers, leachate collection	50%	–
Ground-water monitoring	30	30%
Best management practices	15	–
Corrective action	5	70

Costs per Landfill

The five options impose significantly different costs on units depending on the design requirements. Table 8 presents the annualized combined costs on a per-unit basis. Median costs range between $31,000 and $163,000 under a 10-year post-closure care period and between $43,000 and $177,000 for a 40-year care period. (Note that we could not provide unit cost breakdown for the lower bound). The table also shows maximum costs per landfill, which range as high as $18.6 million per year under Alternative 2. This maximum represents a very large landfill with the most stringent (uniform) design. Minimum costs per landfill across all options are less than $5,000 per year. These minimums reflect small landfills that, due to environmentally stable locations and/or existing state controls, incur few incremental costs.

Table 8

Range of Annualized Costs per Landfill for All Options

Costs Include Corrective Action

	10-Year Post-Closure Care Period[1]		
	Minimum Cost per Landfill	**Median Cost per Landfill**	**Maximum Cost per Landfill**
Final Rule	$1,700	$44,000	$7,170,000
Alternative 1 (Proposal)	2,900	68,000	6,220,000
Alternative 2 (Subtitle C Design)	3,400	163,000	18,300,000
Alternative 3 (Categorical)	1,600	59,000	5,910,000
Alternative 4 (Statutory Minimum)	1,100	31,000	2,070,000
	Upper Bound (40-Year Post-Closure Care Period)[2]		
	Minimum Cost per Landfill	**Median Cost per Landfill**	**Maximum Cost per Landfill**
Final Rule	$1,900	$54,000	$7,300,000
Alternative 1 (Proposal)	3,000	72,000	6,270,000
Alternative 2 (Subtitle C Design)	4,500	177,000	18,600,000
Alternative 3 (Categorical)	1,900	69,000	6,040,000
Alternative 4 (Statutory Minimum)	1,300	43,000	2,140,000

[1] Post-closure care is set at 30 years for Alternative 1; no adjustments for additional waste diversion or regionalization.

[2] No adjustments for additional waste diversion or regionalization.

Cost per Ton

The incremental cost per ton (CPT) for landfills across the regulatory options is summarized in Table 9. Costs at the median landfill range between $11.60 and $13.80 for the final rule, depending on the length of the post-closure care period. The final rule median cost is about 47 percent higher than the statutory minimum, but lies below all of the other options and is significantly lower than the Subtitle C design in all scenarios. The average CPT is presented here as the national average. Note that the median is significantly higher than the national average, reflecting the large number of small landfills with comparably high costs.

Table 9

Range of Incremental Landfill Cost per Ton for All Options

Costs Include Corrective Action

	10-Year Post-Closure Care Period[1]			
	Minimum Cost per Ton	**Median Cost per Ton**	**Average Cost per Ton**[2]	**Maximum Cost per Ton**
Final Rule	$0.58	$11.60	$4.70	$48.70
Alternative 1 (Proposal)	0.82	16.70	4.50	47.90
Alternative 2 (Subtitle C Design)	0.68	39.60	13.20	85.60
Alternative 3 (Categorical Approach)	0.58	14.90	5.60	43.90
Alternative 4 (Statutory Minimum)	0.15	7.90	3.40	111.00

	Upper Bound (40-Year Post-Closure Care Period)[3]			
	Minimum Cost per Ton	**Median Cost per Ton**	**Average Cost per Ton**[2]	**Maximum Cost per Ton**
Final Rule	$0.67	$13.80	$5.00	$52.80
Alternative 1 (Proposal)	0.85	17.80	4.70	49.40
Alternative 2 (Subtitle C Design)	0.79	41.50	13.70	92.30
Alternative 3 (Categorical Approach)	0.67	17.00	5.90	48.10
Alternative 4 (Statutory Minimum)	0.23	9.40	3.60	113.00

[1] Post-closure care is set at 30 years for Alternative 1; no adjustments were made for additional waste diversion or regionalization.

[2] Average cost per ton is calculated by dividing the total annualized costs by the total tons disposed per year.

[3] No adjustments for additional waste diversion or regionalization.

Economic Impact Results

Municipalities

Because more than 80 percent of MSWLFs are publicly owned, the rule has a major effect on municipalities. We first considered the potential impact on municipal budgets as a result of having to comply with the rule. The assessment is difficult because of the uncertainty about how communities will comply with the budget implications of the rule (e.g., raise taxes, cut other services). To obtain some estimate of potential effects, we measured incremental compliance cost as a percentage of community expenditures (CPE). These impact measures focus on the upper-bound (most conservative) cost scenarios for the regulatory options. Communities are classified as experiencing severe impacts if the CPE exceeds 1 percent of total budget. The number of communities and populations with severe impacts are shown in Table 10. All of the options result in CPE greater than 1 percent for at least 14 percent of the jurisdictions.

Table 10

Municipalities and People Affected by Regulatory Costs Exceeding 1 Percent of Current Community Expenditures

(40-year post-closure care period)

	Municipalities		People	
Option	**Number**	**Percentage of Total**	**Number (millions)**	**Percentage of Total**
Final Rule	7,822	27.0%	23.6%	10%
Alternative 1 (Proposal)	8,515	29.3	22.3	9
Alternative 2 (Subtitle C)	18,581	64.0	68.5	29
Alternative 3 (Categorical)	10,319	35.6	27.9	12
Alternative 4 (Statutory Minimum)	4,168	14.4	10.1	4

Note: Regulatory costs reflect the upper-bound scenario. Shifts to recycling, combustion, and regional landfills will mitigate the impacts shown here.

There are significant differences in costs as a share of the total community budget for the various options. Table 11 shows CPE greater than 1 percent for three regulatory options under two scenarios. The upper bound reflects a 40-year post-closure care period. The lower bound reflects the drop in communities with severe impacts after waste has shifted to regional landfills (see Chapter VIII). Under the least protective option – the statutory minimum – costs would exceed the 1 percent impact threshold for 2 to

14 percent of local governments (representing less than 4 percent of the U.S. population). The final rule would result in 4 to 27 percent of communities (2 to 10 percent of the U.S. population) exceeding this threshold. In contrast, under the Subtitle C option, 10 to 64 percent of local governments (6 to 29 percent of the U.S. population) would exceed the 1 percent threshold. These results clearly indicate that the Subtitle C option would impose a dramatic increase in cost of solid waste disposal for a large number of local governments. It is important to recognize that increased regionalization can significantly reduce the impacts, as demonstrated by the ranges in the table.

Table 11

Costs as a Percentage of Expenditures

Range of Impacts for Selected Options

	Percentage of Communities with CPE >1 percent		
Regulatory Option	**Lower Bound**	**Upper Bound[1]**	**Maximum CPE**
Final Rule	4%	27%	7.2%
Alternative 2 (Subtitle C Design)	10	64	13.1
Alternative 4 (Statutory Minimum)	2	14	19.7

[1] More than 80 percent of the communities with CPEs greater than 1 percent are small communities with populations under 5,000.

The greatest impacts for all options fall on small communities (communities with a population of 5,000 or less); 80 percent of significantly impacted communities are small. However, the percentage of the total U.S. population residing in these communities is much smaller than the percentage of communities affected. In addition to being small, two out of three communities facing an incremental CPE greater than 1 percent from the final rule are in rural areas (i.e., have population densities lower than 40 people per square mile). Further discussion on impacts to small communities is discussed in the next section of this Executive Summary.

The magnitude of the CPE is also an important indicator of impacts. The maximum CPE under the statutory minimum is high (nearly 20 percent of the community's expenditure); the maximum CPE for the Subtitle C option is 15 percent. The maximum CPE for the final rule is substantially lower, 7 percent. The high CPE maximum under the statutory minimum primarily results from the high corrective-action costs anticipated at some sites under this option.

Several factors will tend to mitigate the actual impact of the regulatory options on communities. One important factor is the relatively small proportion of the municipal budget that is usually devoted to MSW disposal. Data from EPA's MSWLF survey indicate that the typical community currently spends approximately 0.5 percent of its budget on solid waste disposal. Although CPE greater than 1 percent indicates that

MSW disposal expenditures may double in many communities after regulation, these expenditures will still represent less than 2 percent of the total municipal budget in most communities. For comparison, the average community in the United States currently spends 36 percent of its budget on education, 5 percent for police protection, 3 percent for sewage disposal, 2 percent for fire protection, and 1 percent for sanitation services other than sewage, including solid waste collection and disposal and street cleaning. Although it may be difficult for communities to cope with large percentage increases in MSW disposal costs in the short run, once the initial adjustment is made, these costs should be easier for communities to absorb because they represent a very small portion of communities' total budgets.

In addition, these impacts are estimated using costs that assume 40 years of post-closure care and no shifts of waste to larger, regional landfills or diversion of waste to recycling or combustion. Table 11 indicates that regionalization alone could significantly reduce costs and impacts. Other factors that would decrease incremental costs and impacts but have not been included in the analysis are better siting of future landfills to take advantage of the performance standard, and state landfill requirements that are already in place. Finally, the analysis has not accounted for the small-community exemption that would provide relief to approximately 10 percent of significantly impacted communities.

The impact analysis also evaluated the financial capability of municipalities that will incur the projected regulatory costs. The results of the analysis indicate that the practicable capability of some communities to implement the regulations will be strained. Of the communities with weak financial capability, 18.2 percent will have high CPE (over 1 percent) for complying with the final rule. These communities, in particular, are most likely to experience difficulties affording the revisions to the criteria. Of the communities with CPE greater than 1 percent, 16 percent have weak financial capabilities, 46.4 percent have average financial capabilities, and 30 percent have strong financial capabilities.

Households

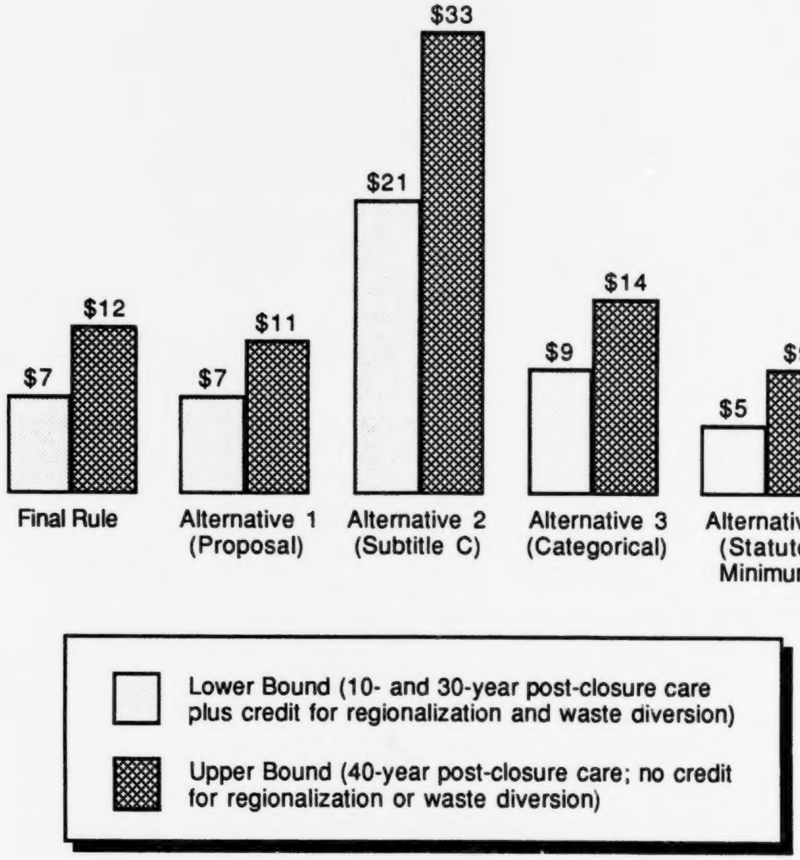
EPA looked primarily at two community-specific measurements – cost as a percentage of median household income (CPMHI) and cost per household (CPH) – to assess impacts on households. The analysis classifies household impacts as severe if the cost per household exceeds $220 (1 percent of national median household income) or if the cost as a percentage of median household income exceeds 1 percent. EPA also computed national cost per household estimates for each option (national costs divided by total U.S. households).

Figure 1 summarizes the national impacts of the regulatory alternatives on households. With the upper-bound estimates, the national average cost per household ranges from $9 under Alternative 4 to $33 under Alternative 2. The final rule falls between those values for the alternatives and averages $12. Using the lower-bound estimates, national average cost per household ranges from $5 to $21.

Figure 1

Average Incremental Cost per Household Nationwide

(dollars per year)



The community CPH threshold level for severe impacts on households, $220 per year (1 percent of the national median household income), is exceeded only under Alternative 2 and Alternative 4; 1 percent of communities exceed the threshold under Alternative 4.

The final rule does not surpass even a moderate impact threshold for individual households (defined for this analysis as an incremental increase in household costs greater than $100 per year, or roughly $8 per month) in 99.9 percent of the communities (see Table 12). In comparison, the Subtitle C approach would surpass the moderate impact threshold for households in 20 percent of the communities; and the statutory minimum would have moderate impacts on households in 6 percent of the communities due to high clean-up costs.

Table 12

Cost per Household (CPH) per Year

40-Year Post-Closure Care Period; CPH for Selected Options

Regulatory Option	Percentage of Communities with CPH >$100	Maximum CPH
Final Rule	0.1%	$175
Alternative 2 (Subtitle C Design)	20.1	305
Alternative 4 (Statutory Minimum)	5.9	362

Cost per household as a percentage of median household income (CPMHI) is relatively low across all of the regulatory options. In our analysis, we used a threshold for significant impacts of 1 percent (the same level used by EPA's construction grants program to identify communities that might experience difficulties affording new wastewater treatment facilities). In our analysis, this threshold was exceeded under all of the options except the proposal. Less than 0.1 percent of communities exceeded the threshold under the final rule, while 1.5 percent exceeded the threshold under the statutory minimum.

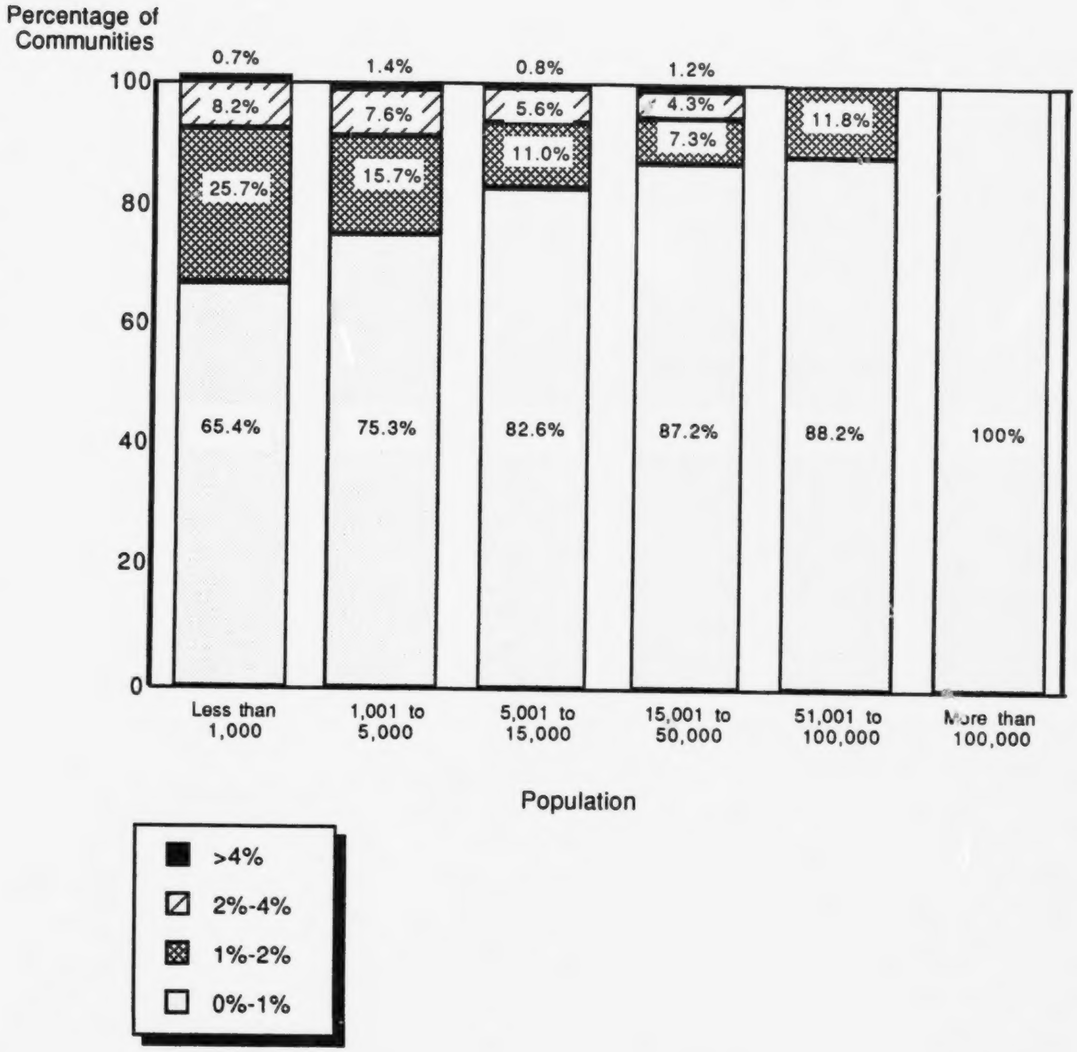
Impacts on Small Communities: Regulatory Flexibility Analysis

As part of the economic impact analysis, we assessed impacts using upper-bound costs on small entities, particularly small governments (defined for the purposes of this analysis as jurisdictions with a population of 5,000 or less), as required by the Regulatory Flexibility Act. The purpose of the analysis was to determine whether a substantial number of small entities incur significant impacts from this regulation. Figure 2 illustrates impact levels across community sizes for one of the three economic impact measures, CPE. This figure shows that differential impacts do occur and that greater impacts fall on small communities.

Figure 2

Incremental Cost as Percentage of Expenditures for Final Rule

By Community Size; 40-Year Post-Closure Care



The significance of those impacts on small communities is less clear, however. First, the final rule imposes significant impacts on fewer small communities than most of the alternatives considered. Only the statutory minimum imposes significant impacts on a smaller share of small communities. However, maximum CPE (Table 11) for the statutory minimum (20 percent) is much higher than for the final rule (7 percent), indicating the severity of potential impact. Second, the impact estimates are based on upper-bound cost estimates, so they reflect 40 years of post-closure care and no shifts to recycling, combustion, or larger, regional landfills. The small landfills serving these communities are also the ones that will respond to higher costs by shifting to larger landfills rather than by replacing their existing small landfills. These small landfills could also benefit substantially from shorter post-closure care periods (an option provided to approved states). Because ground-water monitoring accounts for most of the costs at small landfills, shortening the post-closure monitoring period by 30 years would reduce impacts. Third, these costs still constitute only a small fraction of communities' total budgets. As described earlier, solid waste expenditures typically account for 0.5 percent of budgets in which 2 percent is spent on fire protection and 3 percent for sewage disposal. Finally, we did not consider in this analysis the exemption provided in the final rule for small, remote landfills. That exemption would reduce the impact on approximately 10 percent of small communities.

Based on these mitigating effects, we conclude that the impacts will not be significant for a substantial number of small communities.

Risk and Resource Damage Results

For each of the options, EPA evaluated the human health and environmental benefits resulting from avoided ground-water contamination, developing, where possible, quantitative estimates of these benefits. EPA also carefully considered and qualitatively evaluated other intangible, difficult-to-quantify benefits, such as the existence value of clean ground water to future generations. The risk and resource damage results are summarized below, first for the baseline and then for the regulatory options. In the discussion of results, maximum exposed individual (MEI) risk is the 300-year average individual risk at the well closest to the landfill, as reported in the MSWLF survey. Population risk is expressed as the number of cases caused by releases of one set of new landfills (20 years worth of landfilling) modeled over the 300-year modeling period. Finally, resource damage is the cost expressed in present value (PV) terms to provide water to users whose supply is contaminated by releases from one set of new landfills.

We should emphasize that these two measures of benefits, risk and resource damage, are mutually exclusive and cannot be added together. They are best viewed as alternative measures of benefit that emphasize different aspects of the benefits.

Baseline Risk

MEI risk (i.e., risk to a person exposed at the closest well) in the baseline ranges from zero to about 10^{-4}. About 60 percent of the landfills pose no risk. For approximately 54 percent of the landfills, we simulate zero risk because no drinking-water wells are reported within one mile of the unit (we model risk out to a maximum well distance of about one mile). The remaining 6 percent have no risk because constituents do not reach the nearest well during the 300-year modeling period. A total of 6 percent of the landfills pose actual MEI risks higher than 10^{-5} and a total of 17 percent of the landfills have MEI risk exceeding 10^{-6}. Less than one-tenth of a percent have risks higher than 10^{-4}. This risk distribution can be expected to shift toward higher risks if new wells are placed near existing landfills or if new landfills are sited near existing wells. The estimated number of cancer cases caused in the baseline, over 300 years, by one set of 6,034 new landfills is 23.

Estimated risk is caused entirely by carcinogens, predominantly vinyl chloride; 1,1,2,2-tetrachloroethane; and dichloromethane. We believe that these pollutants are present in MSWLF leachate primarily as a result of disposing household hazardous waste, small quantity generator waste, and industrial solid waste.

Risk tends to be higher in wetter climates, for larger facilities, and at wells closest to landfills. No single factor is responsible for most of the variability; instead, there is a complex interaction among the factors that govern leachate flux and flow through the underlying aquifer.

The baseline of 23 cancer cases caused by one set of replacement MSWLFs is low. Several factors lead to this low estimated cancer incidence. First, more than half (54 percent) of the landfills are sited where no down-gradient drinking-water wells exist within a one-mile radius. These landfills are therefore assumed to present no human health risks. Second, EPA modeled human health risk by using the average population density of well-using residents near MSWLFs (i.e., 1.6 well users per acre). Risk will increase if the population living near landfills increases, as may occur in the future. Third, EPA used median leachate concentrations in the risk modeling. If EPA had used the 90th percentile of leachate concentration in the analysis, the human health risk estimates for 10 percent of the landfills would have increased by, at most, a factor of 10. Therefore, while near-term human exposure to contaminated ground water is clearly a concern for a portion of MSWLFs, greater benefits may result from preventing future human exposure and future loss of ground-water resources.

Baseline Resource Damage

Under the baseline modeling assumptions, the cost to replace contaminated ground water at landfills ranges from zero to more than $4 million (present value or PV) per landfill. About 29 percent of the landfills have no resource damage, indicating that pollutant levels in ground water did not exceed MCLs or taste and odor thresholds during the modeling period. Forty-nine percent of landfills have replacement costs between zero and $200,000. Because ground water in the vicinity of more than half of the landfills is not currently used, most contamination causes resource damage that has relatively low present value. For units where nearby populations currently use ground water, replacement costs can be high. Replacement costs exceed $1 million (PV) for 13 percent of the landfills. The median replacement cost is about $79,000 (PV), and the total resource damage for 6,034 new landfills is about $2.58 billion (PV).

Incremental Benefits: Risk

EPA found that all the regulatory options would achieve roughly similar results for reduction in human health risks caused from drinking contaminated ground water. As indicated in Table 13, both the final rule and the statutory minimum would eliminate approximately 16 cancer cases (70 percent reduction from baseline) occurring over 300 years from releases from one set of replacement landfills identical in size and location to those now operating in the United States. The Subtitle C approach, which imposes the most stringent containment system design on all landfills, would eliminate an additional three cancer cases. The proposal would be less effective than the final rule.

Table 13

Predicted Population Risk Across One Set of Replacement MSWLFs[1]

30-Year Post-Closure Care Period

Regulatory Scenario	Cancer Cases for One Set of Replacement MSWLFs	Reduction of Cases	Percentage Decrease[2]
Baseline	23	N/A	N/A
Final Rule	7	16	68
Alternative 1 (Proposal)	8	15	64
Alternative 2 (Subtitle C Design)	4	19	82
Alternative 3 (Categorical Approach)	5	18	77
Alternative 4 (Statutory Minimum)	7	16	70

N/A = Not applicable.

[1] The population risk is calculated as the total number of cancer cases caused by releases from one set of new landfills (20-year lifespan) modeled over 300 years. They do not represent the total risk of landfilling in perpetuity and, therefore, are not directly comparable to the cost numbers (which represent landfilling in perpetuity) presented earlier in this summary.

[2] Percentage decrease may not reflect the estimates presented in the table because of rounding.

Incremental Benefits: Resource Damage

In looking at long-term protection of both human health and the environment, EPA determined that the preventive options would provide more effective, long-term resource protection than the statutory minimum. Resource damage was estimated as the cost of replacing contaminated ground water with an alternative water supply system. Table 14 summarizes the resource damage estimates for each option. The Subtitle C design and the categorical approach are the most effective at reducing resource damage. The statutory minimum is the least effective.

Table 14

Present Value of Resource Damage[1]
New Landfills Only

(dollars in millions)

	Resource Damage		Net Reduction in Resource Damage	
	10-Year Post-Closure Care Period[2]	40-Year Post-Closure Care Period	10-Year Post-Closure Care Period[2]	40-Year Post-Closure Care Period
Baseline	$2,580	$2,580	–	–
Final Rule	1,240	840	$1,340	$1,740
Alternative 1 (Proposal)	1,290	1,210	1,290	1,370
Alternative 2 (Subtitle C)	690	280	1,890	2,300
Alternative 3 (Categorical)	900	440	1,680	2,140
Alternative 4 (Statutory Minimum)	1,640	1,610	940	970

[1] The avoided resource damage is based on one set of new landfills modeled over 300 years. It does not show avoided resource damage from landfilling in perpetuity and, therefore, is not directly comparable to the cost numbers presented earlier.

[2] Post-closure care period is 30 years for Alternative 1.

In summary, for the benefits measured, the final rule and Alternative 4 have similar impacts on risk reduction. Reductions in the annual number of cancer cases range between 68 and 70 percent for these options. The final rule is the third most effective option at reducing risk and the third most effective at reducing resource damage if post-closure periods are set at 40 years.

Alternatives 2 and 3 are more effective than the final rule and the proposal because the latter two target only the higher-risk landfills (i.e., those with releases exceeding MCLs or MEI risks of 10^{-5}). Alternatives 2 and 3, on the other hand, impose more stringent design standards on all or most of the landfills.

Comparison of Costs and Benefits

The incremental costs of the regulatory options and the reductions in human health risk and resource damage provides one way to compare quantitative aspects of the regulatory options. Additional, non-quantified benefits of the rule (cited earlier in this Executive Summary) have not been incorporated into these measures. Therefore, these comparisons should be viewed as partial measurements of the true effectiveness of the regulatory options.

In order to compare risk reduction with incremental costs, we had to present the results of each analysis in consistent formats. We used total cancer cases caused from one set of new landfills, total present value resource damage caused by one set of new landfills, and total incremental present value cost of one set of new landfills (note that these costs differ from the costs presented earlier for landfilling in perpetuity).

We calculated the present value of costs for all landfills and for the subset of landfills with wells within one mile. The analysis of costs for landfills with wells illustrates a scenario that may occur in the future where virtually all landfills have nearby drinking-water wells. In the current set of surveyed landfills, only 46 percent reported wells within one mile. Both costs (total and with wells) are presented to illustrate the total cost of the options.

The cost estimates generated for this analysis incorporate costs for a range of activities associated with the operation of an MSWLF. Some of these costs will have effects on media other than ground water, but the benefit measures are restricted to ground water. Therefore, in these comparisons, we assigned a higher cost than is appropriate for protecting ground water. The ground-water protection costs dominate the incremental cost estimates, however, so the comparisons are reasonable. Table 15 combines estimates of risk reduction and incremental costs for new units.

Table 15

Costs and Cases Avoided

One Set of New Landfills Only; 40-Year Post-Closure Care Period

		Incremental Cost (present value in millions)	
	Cases Avoided	**All Landfills**	**Landfills with Wells**[1]
Final Rule	15.8	$14,800	$6,560
Alternative 1	14.7	13,000	6,170
Alternative 2	18.8	46,700	21,000
Alternative 3	17.7	18,600	8,910
Alternative 4	16.1	11,800	5,670

[1] These costs are computed only for the fraction of MSWLFs that reported drinking-water wells within one mile of the units.

Incremental costs vary much more than the cases avoided. Some of the alternatives, therefore, impose much greater costs for limited additional benefit. Between Alternatives 3 and 2, for example, landfill owners would incur an additional $28 billion to avoid 1.1 additional cases over 300 years. We should also note that Alternative 4 avoids more cases and costs less than Alternative 1.

Table 16 presents estimates for avoided resource damage and cost at new units. To compare costs with benefits, we calculated the net benefit as reduction in resource damage minus incremental cost. When the net benefit is less than 0, as it is for all the options, costs exceed resource damage benefits. The final rule falls in the middle of the range of net benefits: The statutory minimum and the proposal have higher net benefits (although still negative). The rank ordering of the options is not affected by the post-closure care period.

Table 16

Net Benefit of Resource Damage Improvements

(present-value dollars in millions)

	10-Year Post-Closure Care Period[1]			40-Year Post-Closure Care Period		
	Reduction In Resource Damage	Incremental Cost	Net Benefit	Reduction In Resource Damage	Incremental Cost	Net Benefit
Final Rule	$1,340	$14,000	($12,700)	$1,740	$14,800	($13,100)
Alternative 1	1,290	12,700	(11,400)	1,370	13,000	(11,600)
Alternative 2	1,890	45,500	(43,600)	2,300	46,700	(44,400)
Alternative 3	1,680	17,800	(16,100)	2,140	18,600	(16,500)
Alternative 4	940	11,100	(10,200)	970	11,800	(10,800)

Note: Costs and resource damage estimates assumed 30-year post-closure care for Alternative 1. Estimates do not reflect waste shifts due to regionalization or diversion of waste to recycling or combustion.

Limitations

There are several important caveats to the cost, economic impact, risk, and resource damage results presented in this RIA. Compliance costs are presented across a broad range to reflect some of the flexibility built into the rule plus shifts in waste due to regionalization and diversion from landfills because of increased source reduction, recycling, and combustion. We did not take these factors fully into account in the economic impact, risk, resource damage, or comparative cost and benefit analyses. While facilities may choose designs other than those we modeled, and may continue post-closure care for different durations, the assumptions we used in the RIA are reasonable and reflect reasonable responses to the regulation.

Several additional factors were not examined in the cost and impact analyses, which leads to an overestimate of actual economic impacts. These factors include improved siting of future landfills, which will decrease design costs; exemptions for small, remote landfills; and existing state requirements for municipal landfills (other than for ground-water monitoring, covers, and containment systems that were included in the analysis).

We also have several caveats related to the risk analysis. The risk and resource damage modeling includes considerable uncertainty. The model components that introduce the most uncertainty are those that predict: (1) leachate quality for trace organics, (2) the probability and consequences of containment system failure, (3) the cost and effectiveness of corrective action, and (4) the human health risk resulting from exposure to toxic substances (i.e., the dose-response models). Also, state regulations for MSWLFs were not factored into the baseline.

The risk analysis considers only the current population that is using the ground water as a drinking-water source. Future population growth would increase the risk reduction estimates presented in this RIA. In the future, greater numbers of people and wells may be located near MSWLFs. The resulting increases in risk may be offset if regionalization occurs so that the total number of landfills is reduced. Future population growth and a corresponding increase in solid waste generation that may be land-disposed would also increase compliance costs from the current estimates.

The risks presented in this RIA are limited to those attributable to drinking contaminated ground water. Other risks from MSWLFs were not analyzed (e.g., surface water contamination, subsurface gas, risks to the ecosystem) and are discussed in the report to Congress. Analyzing these risks would result in greater risk reduction than currently estimated. The aggregate costs include some of the controls that would prevent these other forms of risk, although the bulk of the compliance costs are for requirements that serve to protect the ground water from leachate contamination.

One other concern is that risk reduction is only modeled at new units. Additional risk reduction is also likely at existing units. If existing units were included, total risks and resource damage avoided would increase. Also, the modeling period used in the risk analysis is 300 years. Greater risk reduction would be obtained if this period were extended.

Many assumptions, like those discussed above, enter into the risk analysis. Thus, strong reliance on the absolute risk or cost estimates without full realization of the limitations of the analysis should be avoided. In general, relative comparisons of the risk estimates across regulatory options are more appropriate than judgments based on the absolute estimates for a given option.

In addition, several hard-to-quantify benefits of a prevention strategy versus a pollute-and-clean-up strategy were not included in this analysis. Many of these were mentioned earlier in the summary. They include the existence value of ground water, greater protection from prevention versus reliance on ground-water monitoring and clean-up technologies, reduced potential for intergenerational inequities, reduced potential that landfill will become a federal burden, reduced social-welfare impacts (i.e., limiting impacts on property value), avoidance of extremely costly future burdens on financially ill-prepared communities for corrective action, and increased public confidence in landfills.

I. Introduction

The Federal Role in Solid Waste Disposal

Since 1979, the federal government has had criteria (40 CFR Part 257) for classifying solid waste disposal facilities as either sanitary landfills or open dumps. These criteria fall under Subtitle D of the Resource Conservation and Recovery Act (RCRA) and include performance standards for facilities classified as sanitary landfills. Facilities failing to meet the criteria are classified as open dumps and must be closed or upgraded.

The Subtitle D program was designed as a cooperative effort between federal, state, and local governments; the federal role was to establish minimum standards and provide implementation funding to states. This funding was cut off, however, in 1981, when the focus of RCRA turned to hazardous-waste issues primarily under Subtitle C.

The Hazardous and Solid Waste Amendments of 1984 (HSWA) marked the federal government's reentry into the Subtitle D arena. Section 4010 of HSWA requires the Environmental Protection Agency (EPA) to promulgate revisions to the criteria for facilities that may receive household hazardous waste or small-quantity generator waste. The revised criteria must protect human health and the environment and, at a minimum, should:

- Require ground-water monitoring
- Establish criteria for the acceptable location of new or existing municipal solid waste landfills (MSWLFs)
- Provide for corrective action as appropriate

Finally, in developing the criteria, EPA may take into account the "practicable capability" of MSWLF owners and operators.

The Nature of the Problem

The new federal initiative incorporated into HSWA emerged out of Congress's concern over the quantity of hazardous wastes and other dangerous materials that are placed in Subtitle D landfills. These wastes may include household hazardous wastes, small-quantity generator wastes, or illegally dumped hazardous wastes. In addition, 184 of the 850 sites on or proposed for the National Priority List (compiled for the Superfund program as of May 1986) are MSWLFs. EPA has assessed many of these concerns in its Report to Congress, cited in Chapter II of this report.

The Subtitle D universe consists of MSWLFs, industrial landfills, land application units, surface impoundments, and waste piles. EPA has adequate data for regulatory decisions only on MSWLFs, but is gathering data on other Subtitle D facilities, and will assess the need to regulate these facilities as this information becomes available. Thus, this rule-making applies only to MSWLFs. Preliminary survey results indicate that there are 6,034 MSWLFs that may receive household hazardous waste and small-quantity generator waste.

Required Analyses

Executive Order 12291 requires federal agencies to prepare a regulatory impact analysis (RIA) for all major regulations, defined as those expected to have an annualized cost of more than $100 million. Because the modifications to the Subtitle D criteria will entail costs of at least this magnitude, EPA has prepared this RIA.

An RIA typically evaluates regulatory options by considering their costs, effectiveness, economic impact, cost-effectiveness, and, where possible, net social benefits. This RIA performs the evaluation in terms of two charges from Congress: that the revisions protect human health and the environment, and that they consider the practicable capability of MSWLF owners and operators.

To assess the effects on human health and the environment, the incremental effects of regulatory options in reducing human health risk and resource damage are evaluated. Both risk and resource damage measures are based on improvements in ground-water quality. Human health risks are measured in terms of risk to the most exposed individual (the individual exposed at the ground-water well nearest to the landfill) and risk to all exposed people within one mile of the MSWLF. Resource damage is measured as the cost of an alternative drinking water supply to replace the contaminated ground water.

The practicable capability issue is addressed by measuring the incremental costs of the regulatory options and comparing these costs to current total community expenditures, to median household income, and to current MSWLF operating costs. The affordability of each option is measured relative to the financial capabilities of the municipalities that own or use the facilities. Each of these components of the analysis is discussed in more detail in Chapter V.

EPA is also required under the Regulatory Flexibility Act of 1980 to assess whether the regulation will have a "significant economic impact on a substantial number of small entities." If a rule does have such an impact, EPA must then complete a regulatory flexibility analysis (RFA). The entities of primary concern for this proposed rule-making are small governmental jurisdictions, defined in the act as governments of a district with a population of less than 50,000. Since more than half of the MSWLFs are owned by governments of this size, a different cutoff may be appropriate. Alternatives to the size criteria are discussed in Chapter VIII, along with criteria for "significant economic impact" and "substantial number."

Outline of the Report

This report contains ten chapters. Chapter II describes the background for the final rule, which EPA has labeled the "hybrid" approach in the preamble to the regulation. All references to the final rule in the RIA describe this hybrid regulatory approach. We describe the current regulatory requirements for landfills (the term "landfill" is used interchangeably with MSWLF); the characteristics of landfills that will be subject to new regulatory requirements (including their size, location, and ownership); and the design and operating practices of existing landfills.

Chapter III presents the regulatory options, including the final rule, the 1988 proposal, and three other regulatory alternatives. The final rule was developed in response to the August 30, 1988, proposal and the ensuing public comments.

In Chapters IV through VI we present the methodology for the cost analysis, economic impact analysis, and the risk and resource damage analysis. Chapters VII through IX follow the same organization and present, in turn, the cost results; economic impacts, including the impacts of the options on small entities; and health risks and resource damage for each regulatory option.

Chapter X combines the cost and risk and resource damage estimates.

II. Background

This chapter begins with a summary of existing federal and state regulatory requirements for Subtitle D facilities. The next section describes those characteristics – number, size, remaining life, ownership, location, and operating practices – of existing MSWLFs which could be affected by new regulations promulgated by EPA.

Current Regulatory Requirements

Federal Regulations

As described in Chapter I, EPA promulgated the Subtitle D criteria (40 CFR Part 257) in 1979. These rules were corrected and amended in the *Federal Register* twice in 1979 and again in 1981. The final rule addresses these criteria as they apply to MSWLFs. The Part 257 criteria set standards in eight areas: floodplains, endangered species, surface water, ground water, land used for production of food chain crops, disease, air, and safety. The level of detail varies significantly among these eight areas of concern.

Floodplains

Existing EPA rules prohibit the siting of municipal landfills in floodplain areas where the landfill would restrict the flow of a 100-year flood. (Such a flood has a magnitude that occurs only once every 100 years.) Landfills should not reduce the temporary storage capacity of a floodplain or result in a washout of solid waste that presents a hazard to humans, wildlife, or land and water resources.

Endangered Species

Neither facilities nor practices may cause or contribute to the harassment, wounding, capture, trapping, or killing of endangered species. Neither facilities nor practices may harm the habitats of endangered species.

Surface Water

Facilities must comply with Sections 404, 208, and 402 (the National Pollutant Discharge Elimination System) of the Clean Water Act.

Ground Water

Facilities may not contaminate ground water beyond the solid waste boundary, unless a court or a state determines that another boundary is an acceptable compliance point and would not contaminate ground water that may be needed for human consumption.

Land Used for Production of Food Chain Crops

The rules set standards for the disposal of waste containing cadmium and polychlorinated biphenyls (PCBs) near or on land used for the production of food chain crops. The criteria set maximum pH and cadmium levels for soil and waste containing cadmium and a cadmium concentration limit for waste applied to land used for the production of certain crops. These limits apply to facilities applying solid waste within one meter of land used for producing food chain crops.

Disease

Part 257 prohibits the operation of a facility "unless the on-site population of disease vectors is minimized through the periodic application of cover material or other techniques as appropriate so as to protect public health" (Part 257.3-6). The same section sets standards for facilities that dispose of sewage sludge and that pump septic tanks.

Air

The criteria prohibit violation of requirements developed under Clean Air Act State Implementation Plans and, with few exceptions, open burning.

Safety

The safety requirements address four potential problem areas: explosive gases, fires, bird hazards to aircraft, and public access. The agency rules set a cap at 25 percent of the lower explosive limit of gas in facility structures and a cap at the lower explosive limit at facility boundaries. In addition, no facility or practice may pose a fire hazard. Part 257.3-8 prohibits the siting of facilities within 10,000 feet of airports serving turboprop aircraft or within 5,000 feet of airports serving piston-only aircraft. Finally, the criteria prohibit uncontrolled public access to landfills, because such access may cause a health hazard.

HSWA

Section 4010 of HSWA required EPA to prepare a report to Congress by November 8, 1987, indicating whether the existing criteria were adequate to protect health and the environment from ground-water contamination and whether additional enforcement authorities were needed. Section 4010 also required EPA to revise the existing criteria by March 31, 1988, for facilities that may receive hazardous household waste or small-quantity generator waste.

Section 4005 of HSWA required all 55 states and territories, not more than 18 months after the promulgation of revised criteria, to "adopt and implement a permit system or other program of prior approval and conditions" to ensure that units receiving household hazardous waste or small-quantity generator waste comply with the revised criteria. All programs are to be approved by the Administrator.

State Requirements

State regulatory requirements can be divided into seven areas: permitting and administrative requirements, design criteria, operation and maintenance standards, location standards and restrictions, monitoring requirements, closure and post-closure care requirements, and financial responsibility. Data on these requirements were collected through an EPA review of state Subtitle D requirements.[1] EPA's findings are summarized below for each of the seven areas.

Permitting and Administrative Requirements

Fifty-four (of 55) states and territories require the issuance of a license or permit before MSWLFs can operate. According to the MSWLF survey, nearly 93 percent of landfills active in 1986 had permits or licenses. The Report to Congress indicated that approximately 82 percent of municipal landfills included in an earlier census has permits or licenses. The majority of permit requirements relate to soil conditions, ground-water information, and surface-water information. Less than a third of the states and territories have requirements concerning total acreage, the life of the facility, and future use of the property.

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[1] U.S. Environmental Protection Agency, Office of Solid Waste, *Final Report on Solid Waste Disposal in the United States*, Report to Congress, Vol. 2 (October 1988), EPA 530-SW-88-011B, Chapter 5 and Appendix D, Tables D-1 through D-6.

Design Criteria

Fifty states and territories have standards that require the owner and/or operator to control the generation, storage, collection, transportation, processing and reuse, and disposal of waste in a safe, sanitary, esthetically acceptable, and environmentally sound manner. Information needed to evaluate MSWLF design and operating conditions according to these criteria is generally required on the permit application.

Leachate management is mandated by 27 states and territories, although design specifications vary widely. Twenty-four states have promulgated liner design specifications.

Of the 55 governments studied, 31 require a system for the control of decomposition gas. Many use the lower-explosive-limit requirements derived from the federal criteria. Others require a venting system or odor regulation.

Water run-on and run-off control systems are mandated by 42 states and territories. Criteria in this category require owners and operators to provide adequate drainage to minimize erosion and prevent ponding.

Operation and Maintenance Standards

Operation and maintenance standards, which can be either design or performance standards, are of six types: waste management, leachate controls, gas controls, cover, safety, and miscellaneous other. Minimum operation and maintenance standards are established in 52 states and territories. Waste management standards, although usually very general, are required by all states. Thirty-six states and territories have imposed leachate control standards.

Each state requires compliance with applicable surface- or ground-water quality standards. Twenty-two states require that explosive-gas concentrations be controlled to prevent a hazard. Daily cover of sites is mandatory in most states, although sites handling only inert wastes are frequently exempted. State cover requirements often depend on the population served and amount and type of waste received. The definition of cover also varies, often being just soil or earth but sometimes including criteria such as good compactability, ability to provide an adequate seal, and ability to provide adequate protection against the infiltration of water.

With few exceptions, states require compliance with the federal safety criteria pertaining to open burning, fire control, and access limitations. Other operation and maintenance criteria include standards for drainage, litter, disease vectors (in California), dust, odors, and noise.

Location Standards and Restrictions

Forty-four states and territories have established location standards and restrictions relating to environmentally sensitive areas and setback distances. Specifically, 36 states and territories have floodplain protection standards and 39 states specify minimum distances to man-made or natural structures, such as airports, utility lines, ground waters, and surface waters. Location restrictions in critical habitat areas are specified by 16 states and territories. Only five states impose restrictions in geologically sensitive areas or in areas with particular soil conditions.

Monitoring Requirements

Thirty-eight states and territories require monitoring of ground water. Of these, 20 also require leachate monitoring. Three states and territories have leachate monitoring requirements without ground-water requirements. Ten states and territories require surface-water monitoring. None of the states have air monitoring requirements.

Closure and Post-Closure Care Requirements

Almost all states and territories have some requirements for closure or post-closure. Forty-four have both types of requirements. Seven have only closure requirements.

Financial Responsibility

Twenty of the 44 states and territories with closure and post-closure requirements require some type of financial assurance.

Summary

The implementation of landfill requirements varies greatly among states and territories. Although almost all states and territories have permit requirements, just over half of all Subtitle D landfills are actually permitted, indicating that the scope, stringency, and enforcement of these requirements differ. States tend to address many common concerns through their standards and requirements for design, operation and maintenance, location, monitoring, and closure. The most common standards relate to run-on and run-off control, waste management, leachate control, cover and safety controls, floodplain protection, ground-water monitoring, and closure and post-closure. Not all states, however, address each of these issues in their requirements and standards.

Characteristics of Affected Landfills

This section summarizes the key characteristics of MSWLFs: their number, size, remaining life, ownership, location, and operating practices.

Number

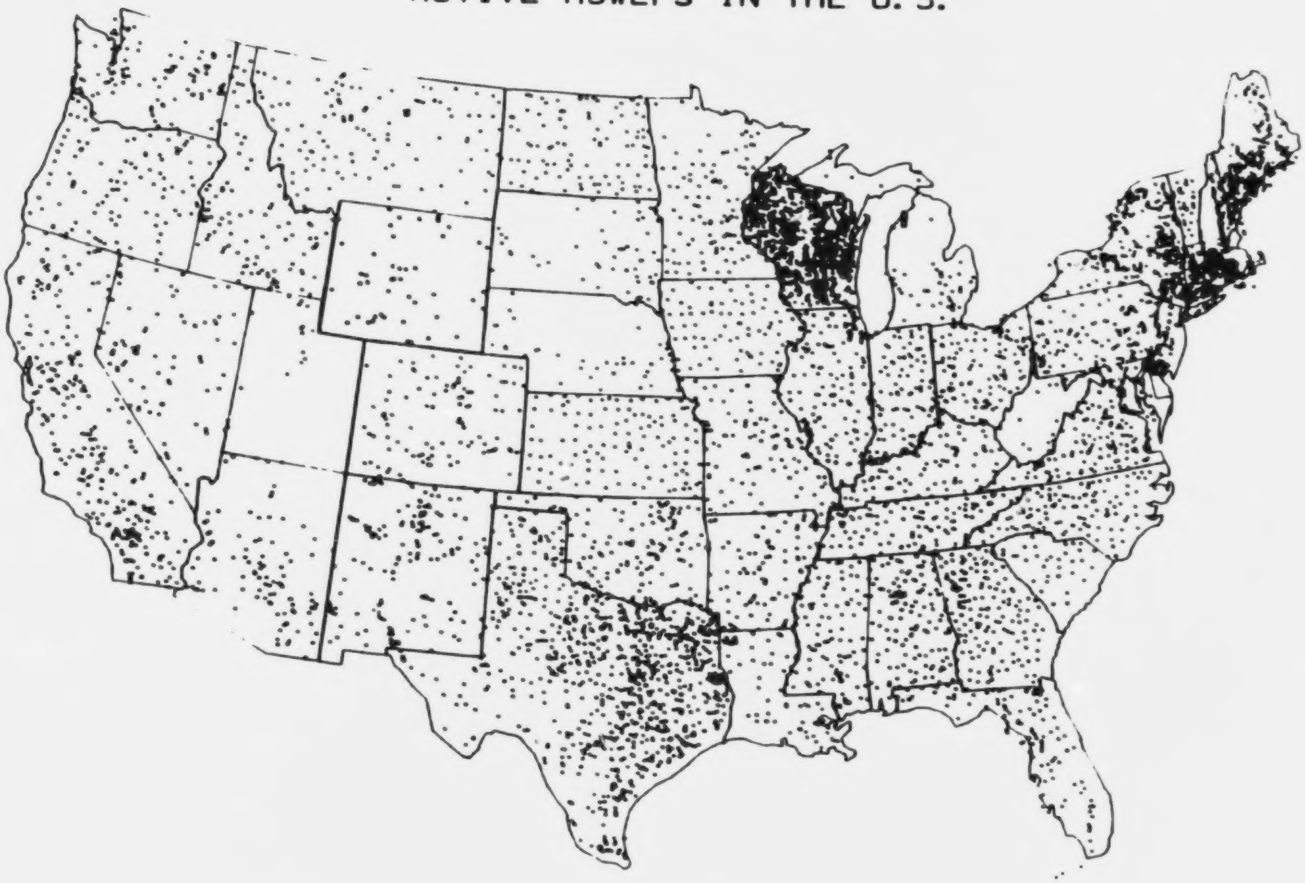
EPA estimates that 6,034 municipal solid waste landfills operate in the United States and its five territories. This estimate is based on results from EPA's 1986 Solid Waste (Municipal) Landfill Survey.

Figure II-1 and Table II-1 show the distribution of MSWLFs across the country and by state. (The number of MSWLFs in each state in Table II-1 is drawn from EPA's earlier MSWLF state census, *Census of State and Territorial Subtitle D Nonhazardous Waste Programs,* which found 7,645 landfills. However, preliminary results from the 1986 survey, which found 6,034 landfills, are regarded as more reliable. The estimates in the table are provided to indicate the distribution across states.) Figure II-1 indicates that landfill locations are most highly concentrated in a few northeastern states (including New England and parts of New York) and in Wisconsin. Wisconsin, which has only 2 percent of the nation's population, has more than 12 percent of all MSWLFs, roughly the same number as Texas and more than twice as many as California, New York, or Maine. Together, these five states – Wisconsin, Texas, California, New York, and Maine – account for about 37 percent of all municipal solid waste landfills.

EPA's 1986 MSWLF survey of 1,250 facilities provides the best recent data on the size, technical characteristics, remaining life, and ownership patterns of municipal landfills. This analysis used results for the 701 active landfills that initially responded to the survey. EPA assigned statistical weighting factors to scale up the responses and estimate the characteristics of all 6,034 landfills. Thus, the data presented below show the estimated size, technical characteristics, remaining life, and ownership patterns for all municipal landfills, not just the 701 initial respondents.

Figure II-1

ACTIVE MSWLFs IN THE U.S.



Note: Because of missing data, only 6,800 of the 7,654 census landfills are plotted. Plotted locations do not match this map exactly.

Source: U.S. EPA State Landfill Census, with landfill location descriptions converted by DPRA, Inc., into latitudes and longitudes.

Table II-1

Distribution of Landfills

State	State Census		State	State Census	
	Number	Percentage		Number	Percentage
AK	179	2.3%	NC	124	1.6%
AL	122	1.6	ND	100	1.3
AR	84	1.1	NE	43	0.6
AZ	96	1.3	NH	70	0.9
CA	401	5.2	NJ	73	1.0
CO	122	1.6	NM	213	2.8
CT	91	1.2	NV	107	1.4
DE	3	0.0	NY	304	4.0
FL	126	1.6	OH	153	2.0
GA	198	2.6	OK	132	1.7
HI	20	0.3	OR	127	1.7
IA	83	1.1	PA	144	1.9
ID	95	1.2	RI	11	0.1
IL	168	2.2	SC	80	1.0
IN	98	1.3	SD	55	0.7
KS	124	1.6	TN	120	1.6
KY	107	1.4	TX	935	12.2
LA	93	1.2	UT	112	1.5
MA	203	2.7	VA	147	1.9
MD	42	0.5	VT	70	0.9
ME	294	3.8	WA	118	1.5
MI	58	0.8	WI	933	12.2
MN	105	1.4	WV	141	1.8
MO	106	1.4	WY	78	1.0
MS	108	1.4			
MT	129	1.7	Total	7,645	100.0%[a]

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[a] Does not add to 100 percent because of rounding.

Source: EPA State Landfill Census.

Table II-2

Size Distribution of Municipal Landfills

Model Size Waste Quantity (TPD)	Actual Size Range (TPD)	Number	Percentage[1]
10	1-17.5	3,093	51.3%
25	17.6-50	1,028	17.0
75	51-125	791	13.1
175	126-275	442	7.3
375	276-563	334	5.5
750	564-1,125	189	3.1
1,500	>1,125	157	2.6
Total		6,034	100.0%

---

[1] Total does not add to 100 percent because of rounding.

Source: EPA Municipal Landfill Survey.

Size

Table II-2 summarizes the estimated size distribution of municipal solid waste landfills. This table shows the number of landfills in seven size categories, measured in tons per day (TPD) of waste handled. These size categories are used later in this report to estimate the unit costs of regulatory options. As the table indicates, most landfills (51 percent) handle less than 17.5 TPD, 17 percent handle between 17.6 and 50 TPD, and 32 percent handle more than 51 TPD.

The size distribution of landfills has important implications for the economic impacts of the regulatory options. Many of the regulatory options entail requirements whose costs achieve economies of scale as the size of the unit increases. Thus, in most cases, the regulatory costs per ton of waste handled are lower for larger MSWLFs than for smaller ones.

Table II-3 shows the amount of waste handled by facilities in each size category. Whereas the number of facilities decreases in each subsequently larger size category, the total amount of waste increases. For example, an estimated 157 facilities (2.6 percent of the total) are in the largest size category, yet these relatively few landfills handle 40 percent of all municipal solid waste that is landfilled.

Table II-2

Size Distribution of Municipal Landfills

Model Size Waste Quantity (TPD)	Actual Size Range (TPD)	Number	Percentage[1]
10	1-17.5	3,093	51.3%
25	17.6-50	1,028	17.0
75	51-125	791	13.1
175	126-275	442	7.3
375	276-563	334	5.5
750	564-1,125	189	3.1
1,500	>1,125	157	2.6
Total		6,034	100.0%

---

[1] Total does not add to 100 percent because of rounding.

Source: EPA Municipal Landfill Survey.

Size

Table II-2 summarizes the estimated size distribution of municipal solid waste landfills. This table shows the number of landfills in seven size categories, measured in tons per day (TPD) of waste handled. These size categories are used later in this report to estimate the unit costs of regulatory options. As the table indicates, most landfills (51 percent) handle less than 17.5 TPD, 17 percent handle between 17.6 and 50 TPD, and 32 percent handle more than 51 TPD.

The size distribution of landfills has important implications for the economic impacts of the regulatory options. Many of the regulatory options entail requirements whose costs achieve economies of scale as the size of the unit increases. Thus, in most cases, the regulatory costs per ton of waste handled are lower for larger MSWLFs than for smaller ones.

Table II-3 shows the amount of waste handled by facilities in each size category. Whereas the number of facilities decreases in each subsequently larger size category, the total amount of waste increases. For example, an estimated 157 facilities (2.6 percent of the total) are in the largest size category, yet these relatively few landfills handle 40 percent of all municipal solid waste that is landfilled.

Table II-3

Waste Handled by Landfill Size Categories

Size Range (TPD)	Total Waste Handled (TPD)	Percentage of Total Waste Handled by All MSWLFs
1-17.5	13,122	1.6%
17.6-50	30,610	3.8
51-125	68,763	8.6
126-275	87,002	10.9
276-563	130,455	16.3
564-1,125	149,218	18.6
>1,125	321,767	40.2
Total	800,937	100.0%

Source: EPA Municipal Landfill Survey.

Remaining Life

For the purposes of this analysis, an important characteristic of a municipal solid waste landfill is the number of years it expects to remain open. Some of the regulatory options analyzed in this report impose fewer requirements on existing facilities, enabling landfill owners and operators to minimize the short-term effects of regulation by operating their existing landfills for as long as possible. Also, the remaining life of a facility affects the allocation of costs between existing and replacement facilities in our analysis. Thus, the number of years that existing MSWLFs expect to remain open affects the impact of the regulatory options discussed later in this report.

Table II-4 indicates the distribution of facilities' remaining lives. Approximately 45 percent of the surveyed MSWLFs plan to close by the end of 1992. Within 15 years of the survey date, or by 2002, almost 75 percent of all MSWLFs currently in operation expect to close. The survey data, which are based only on a sample, indicate that in a few states, including Florida, Massachusetts, New Hampshire, and New Jersey, virtually all currently active municipal solid waste landfills may be closed within 10 years.

Table II-4

Remaining Life of MSWLFs

Years[1] Remaining	Number	Percentage	Cumulative Percentage[2]
0	535	8.9%	8.9%
1-5	2,167	35.9	44.8
6-10	612	10.1	54.9
11-15	1,126	18.7	73.6
16-20	360	6.0	79.6
21-25	231	3.8	83.4
26-35	211	3.5	86.9
36-45	154	2.6	89.4
46-55	144	2.4	91.8
56-75	134	2.2	94.0
76-100	67	1.1	95.1
>100	293	4.9	100.0
	6,034	100.0%[2]	

---

[1] Years of remaining life equals closure year minus 1987.
[2] Total does not equal 100 percent because of rounding.

Source: EPA Municipal Landfill Survey.

It is not possible to use the survey data to examine past trends in total landfill capacity because the data show approximately how much capacity has been added each year, but not how much has been lost. However, recent trends in the number of new facilities opening, summarized in Table II-5, suggest that it is becoming more difficult to site MSWLFs. In the early 1970s, 300 to 400 new landfills were being built each year. During the past 10 years, however, this number has dropped to between 50 to 200 new landfills per year.

While the overall number of landfills has declined in recent years, the quantity of waste disposed has pushed steadily higher, suggesting that the average size of existing landfills is also increasing. The largest additions to overall capacity occurred between 1970 and 1977 as a large number of new landfills opened during that period.

Table II-5

Number of Landfills Opened Each Year: 1970 Through 1986

Start Year	Number of Landfills
1970	381
1971	201
1972	309
1973	334
1974	319
1975	396
1976	262
1977	165
1978	190
1979	144
1980	155
1981	144
1982	206
1983	129
1984	149
1985	57
1986	62

Source: EPA Municipal Landfill Survey.

Ownership

EPA's Subtitle D landfill survey provides data on who owns landfills that accept municipal solid waste. In addition, the *1982 Census of Governments* conducted by the Commerce Department contains information on the characteristics of publicly owned solid waste landfills. This information indicates which types of landfill owners will bear the initial effects of any cost increases stemming from new regulations.

Table II-6 indicates the number of publicly and privately owned landfills. The estimates reflect weighted results for 6,034 landfills from EPA's detailed facility survey. Approximately 81 percent of all MSWLFs are owned by local governments, while 15 percent are privately owned. Not surprisingly, the states and the federal government own relatively few MSWLFs (many of which serve parks, campgrounds, and military bases).

Table II-6

Distribution of Landfill Ownership

Operator	Number of Landfills Owned	Percentage of Total[2]
Local government	4,875	80.8%
State government	82	1.4
Federal government	154	2.6
Privately owned	912	15.1
Unknown[1]	11	0.2
Total	6,034	100.0%

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[1] Ownership data were unavailable.

[2] Total does not add to 100 percent because of rounding.

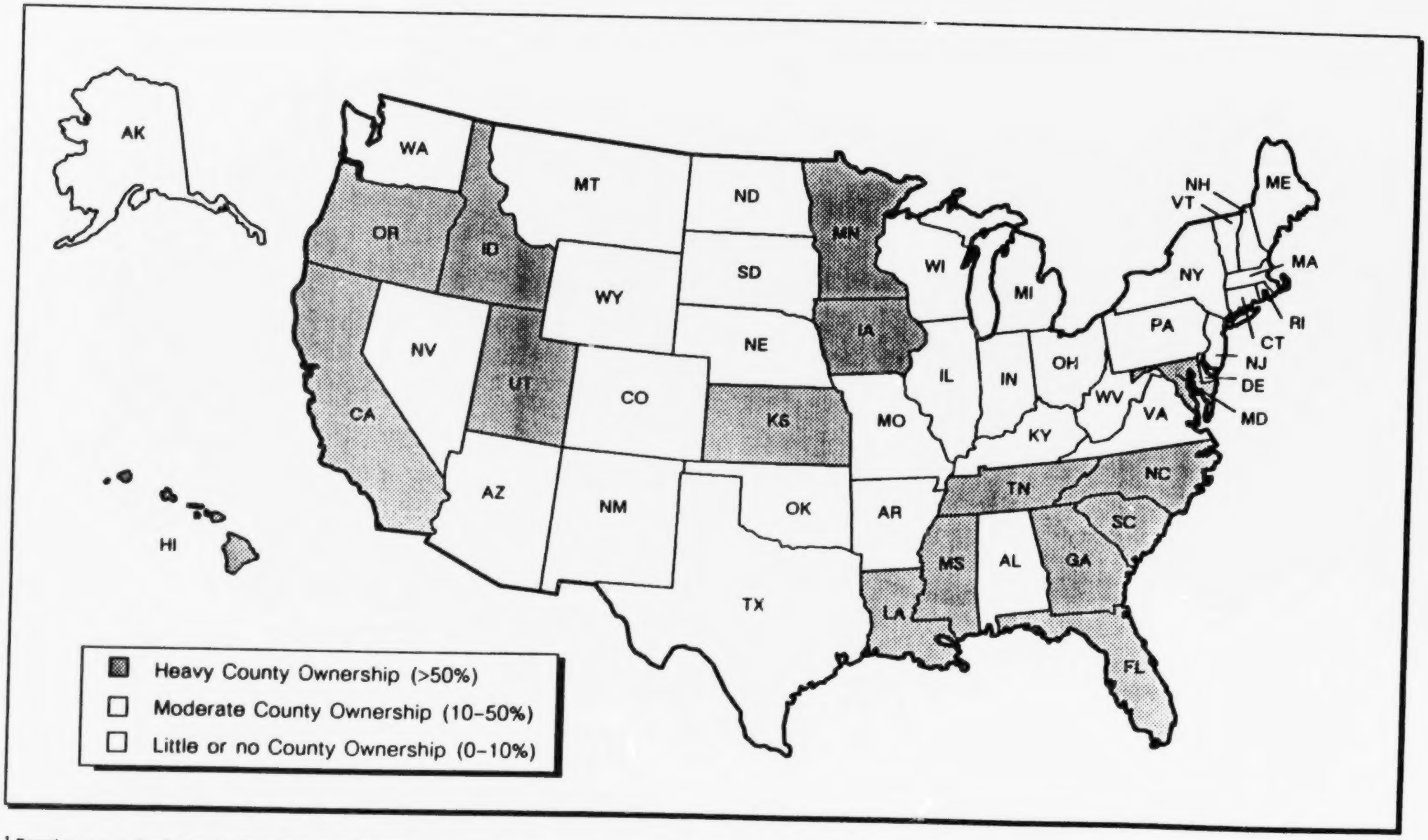
Source: EPA Subtitle D Landfill Survey.

Three types of local governments own landfills: counties, cities, and towns and villages. Preliminary results from EPA's facility survey show that the estimated 4,875 landfills owned by local governments are split fairly evenly among these three types of jurisdictions, with counties owning 36 percent, cities 36 percent, and village or town governments 28 percent. The remaining 1,159 landfills are owned by private contractors, states, special districts, or the federal government. The even distribution of public landfill ownership nationwide masks a great deal of variation in ownership patterns from state to state. Figure II-2 divides states into categories of heavy, moderate, and little or no county ownership of landfills. In New England, New York, and several states in the central part of the country, few, if any, landfills are owned by counties. In most of the southeastern states and in parts of the West, however, the majority of municipal landfills are owned by counties.

Table II-7 presents information from the *1982 Census of Governments* on landfill ownership by local governments. The data examine landfill ownership from a slightly different perspective, focusing on how many local governments actually own landfills. The data are older than those compiled by EPA, and cover only jurisdictions with more than 2,500 people.

Figure II-2

Estimated County Landfill Ownership [1]


AK
HI
WA
OR
CA
NV
ID
MT
WY
UT
CO
AZ
NM
ND
SD
NE
KS
OK
TX
MN
IA
MO
AR
LA
WI
IL
MS
MI
IN
OH
KY
TN
AL
WV
VA
NC
SC
GA
FL
PA
NY
NH
VT
ME
MA
RI
CT
NJ
DE
MD
▨ Heavy County Ownership (>50%)
☐ Moderate County Ownership (10–50%)
☐ Little or no County Ownership (0–10%)

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[1] Based on a sample of approximately 10% of all MSWLFs (i.e., 701 responses to EPA's MSWLF survey).

Source: U.S. EPA Municipal Landfill Survey.

Table II-7

Landfill Ownership Within Local Governments

Population	Number of Governments	Number with Landfills	Percentage
County Government			
500,000+	68	35	51.5%
250,000-500,000	88	36	40.9
100,000-250,000	218	89	40.8
50,000-100,000	374	195	52.1
25,000-50,000	611	291	47.6
10,000-25,000	957	453	47.3
Less than 10,000	725	299	41.2
Total	3,041	1,398	Average 45.9%
Municipal Government			
300,000+	49	24	49.0%
200,000-300,000	23	15	65.2
100,000-200,000	97	45	46.4
50,000-100,000	250	70	28.0
25,000-50,000	525	152	29.0
10,000-25,000	1,253	331	26.4
5,000-10,000	1,515	385	25.4
2,500-5,000	2,078	449	21.6
Total	5,790	1,471	Average 36.4%
Township Government			
100,000+	30	8	26.7%
50,000-100,000	73	14	19.2
25,000-50,000	214	39	18.2
10,000-25,000	702	182	25.9
5,000-10,000	962	217	22.6
2,500-5,000	1,724	323	18.7
Total	3,705	783	Average 21.9%

Source: 1982 Census of Governments.

The data indicate that, nationwide, comparable numbers of landfills are owned by counties, cities, and townships. Larger cities and towns are more likely to own landfills than their smaller counterparts; this is not always true for counties, as the size of the county's population seems to have little effect on whether the county owns a landfill.

The type of ownership is correlated with other important landfill characteristics such as size and operating life. Landfills operated by towns and villages are generally small (over 84 percent handle less than 17.5 TPD), and more likely than not to stay in operation for more than 30 years. County and city landfills are more likely to be moderately sized (i.e., handle between 10 and 275 TPD). Eighty-nine percent of all county landfills and 91 percent of all city landfills fall into this size category. Privately owned landfills are the most likely to be large (31 percent of all private landfills handle more than 275 TPD); however, approximately 37 percent of all private landfills handle less than 17.5 TPD. Private landfills are also somewhat more likely than publicly owned landfills to operate for less than 20 years.

Location of Existing MSWLFs

The discussion of location focuses on areas containing landfills that may be restricted or banned under the final rule. EPA has considered adding location restrictions for six location categories: sites near airports, 100-year floodplains, wetlands, fault areas, seismic impact zones, and unstable areas (e.g., subsidence-prone areas, karst terranes, or landslide-susceptible areas). Existing landfills operating in unstable areas may be required to cease operations, new landfills are restricted from locating in fault areas, and landfills cannot be sited in wetlands. For both the unstable area and wetlands restrictions, landfills in states with approved programs have additional flexibility in complying with the restrictions; landfills in approved states can site in wetlands if they offset any unavoidable impacts on wetlands, and those in unstable areas can operate for more than five years if no practicable alternative exists and there is no danger to human health and the environment. Finally, landfills in seismic impact zones or new units in unstable areas must be designed to resist or mitigate adverse effects from seismic events or instability. The following discussion also covers some of the technical characteristics of MSWLFs, such as the proximity of existing landfills to drinking-water supplies, and subsurface soil type.

Several different sources of data were used to assess the locations of existing MSWLFs. Information on which jurisdictions contain active seismic faults was obtained from Subtitle C (40 CFR Part 264, Appendix VI). Information on the number of MSWLFs located in floodplains, wetlands, and karst areas, as well as data on soil type and proximity to drinking-water supplies, was obtained from preliminary results from the MSWLF facility survey. Note that some inaccuracies in these characterizations may result because they are based on extrapolations from a limited survey sample.

Controlled Areas

Nationwide, about 35 percent of all existing MSWLFs are located in counties that contain seismic faults that have been active within the Holocene period. However, we have no way of knowing how close the landfills in these counties are to the active faults. Four of the five states bordering the Pacific Ocean (Alaska, California, Hawaii, and Washington) have a majority of their MSWLFs located in counties with active fault zones. The six New England states and New York also have a majority of their landfills in this category, as do South Carolina and Tennessee. Many of the states bordering the Rocky Mountains, the Sierra Nevada, and the Mississippi River have at least 25 percent of their landfills in counties where there has been recent seismic activity. However, the two states with the largest number of MSWLFs, Wisconsin and Texas, have no landfills near active fault zones.

About 13 percent of all MSWLFs are located, at least in part, within the 100-year floodplain (see Figure II-3). Six percent are located in areas classified as wetlands, such as swamps, marshes, and bogs (see Figure II-4). An estimated 4 percent of all MSWLFs are located in karst terrane, an area characterized by sinkholes, caverns, and other formations caused by the action of surface and ground water in porous limestone (see Figure II-5).

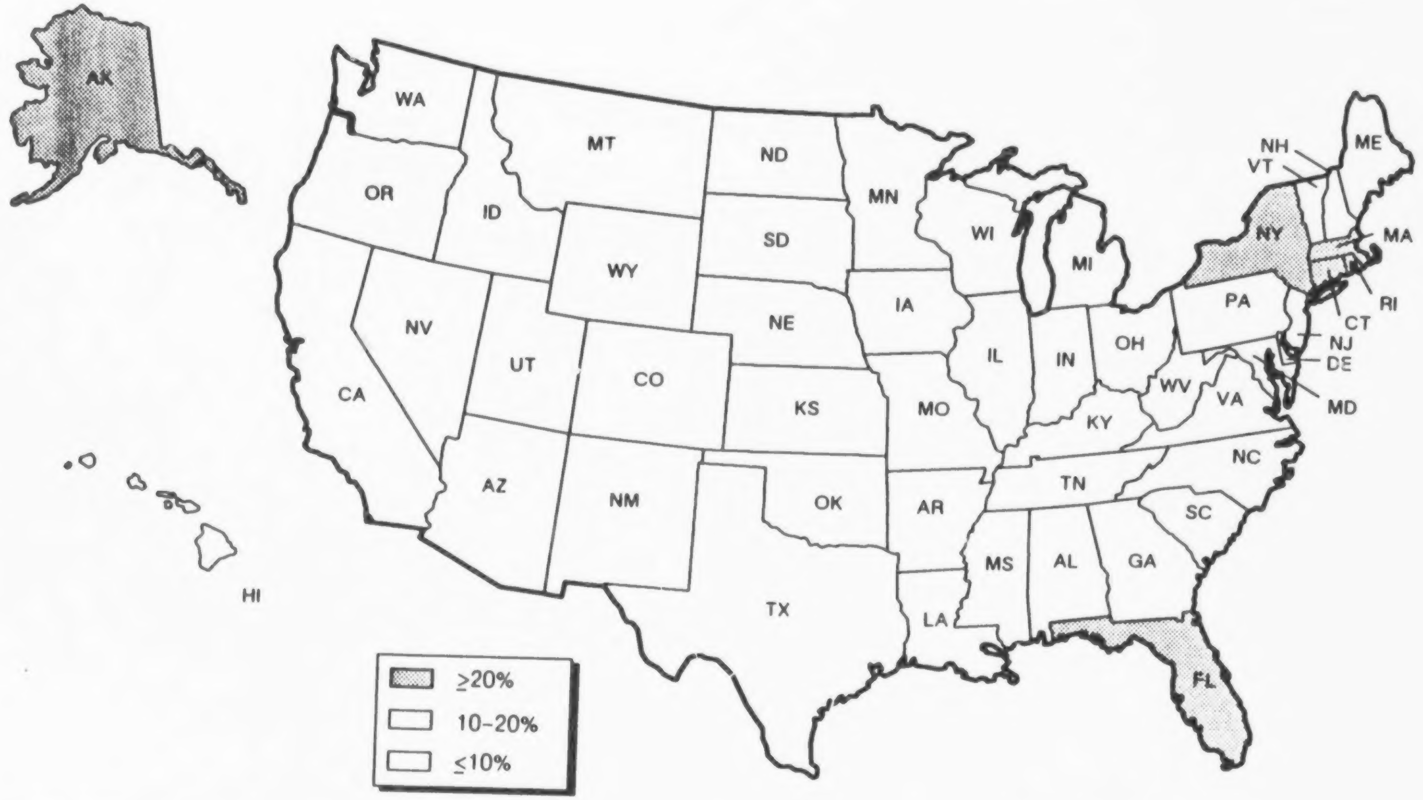
Landfills located in sensitive areas such as these are a concern because the location may cause the failure of containment mechanisms for contamination (e.g., earthquakes or floods may destroy liners or covers designed to minimize contaminant migration) or because contamination may be particularly difficult to clean up (e.g., in wetlands or karst terrane).

Proximity to Drinking Water Supplies

Another location characteristic that increases the potential for human exposure to contamination is proximity to downgradient wells, streams, or reservoirs that are used for drinking-water supplies. Preliminary results from the MSWLF survey indicate that an estimated 46 percent of all MSWLFs are within one mile of a public or private well that provides drinking water, and approximately 6 percent are within one mile of a river or stream that serves as a drinking-water supply.

Figure II-4

Estimated Percentage of Landfills Located in Wetlands [1]



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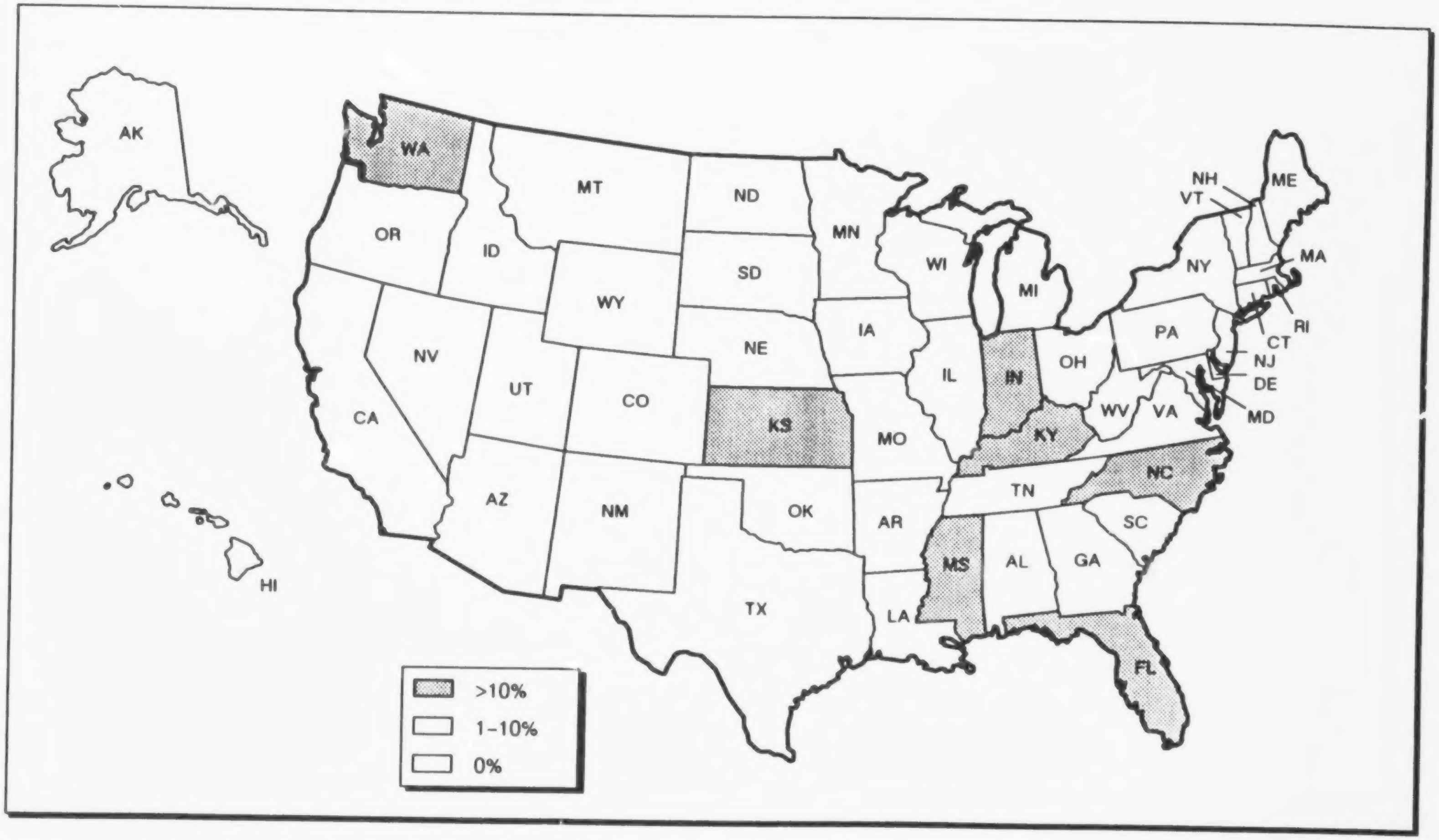
[1] Based on a sample of approximately 10% of all MSWLFs (i.e., 701 responses to EPA's MSWLF survey).

Source: U.S. EPA Municipal Landfill Survey

Figure II-5

Estimated Percentage of Landfills Located in Karst Areas [1]



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[1] Based on a sample of approximately 10% of all MSWLFs (i.e., 701 responses to EPA's MSWLF survey).

Source: U.S. EPA Municipal Landfill Survey.

Environmental Setting

Hydrogeologic characteristics of landfill location play a major role in determining whether any leachate generated at a site eventually reaches the ground water. One characteristic that affects contaminant migration and health risk is the thickness of the unsaturated zone that separates the bottom of the landfill from the aquifer (see Chapter VII). At approximately half of all MSWLFs, the average reported distance from the bottom of the landfilled wastes to the seasonal high-water table is 20 feet or less. At 11 percent of the MSWLFs, there is no separation between landfilled wastes and the seasonal high-water table.

Another factor that influences the rate of contaminant migration is the type of soil in the unsaturated zone. The least permeable type of soil is clay. According to the data compiled through EPA's MSWLF survey, the predominant subsurface soil type for approximately 58 percent of all MSWLFs is either clay or sandy clay.

Operating Practices of MSWLFs

Information on the operating practices of MSWLFs was obtained from the MSWLF survey. This section summarizes the types of waste handled by MSWLFs; it also describes the monitoring practices of existing facilities.

Most MSWLFs accept primarily household waste. At 76 percent of all MSWLFs, over half of the waste received comes from households. MSWLFs accept waste from a variety of other sources, however, including commercial establishments (offices, restaurants, and retail stores), very small quantity generators (VSQGs) of hazardous wastes, and producers of other industrial waste, construction and demolition waste, and sewage sludge. Of these other types of waste, by far the most common types accepted are commercial waste and construction and demolition waste. For most facilities, commercial waste represents at least 15 percent of the waste stream and construction and demolition waste at least 5 percent. Table II-8 shows the percentage of facilities handling each waste type and the percentage whose waste flow consists primarily of each waste type.

Table II-8

Landfills Handling Each Waste Type

Waste Type	Percentage Handling Receive at All[1]	Percentage Handling Accounts for More than 50 Percent
Household	100%	76%
Very small quantity generator	21	0
Commercial	91	4
Industrial	47	0
Demolition/construction	80	0
Sludge	35	0

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[1] Percentage is calculated only for those landfills that responded to the question. For example, only 47 percent answered the question about sludge and 35 percent of those (16 percent of all surveyed landfills) accepted sludge.

Source: EPA Municipal Landfill Survey.

Many landfills did not estimate the amount of VSQG or industrial waste they handle. Of the approximately 30 percent of the facilities that indicated whether they handled VSQG waste, most facilities reported handling no VSQG waste. For those facilities that did report handling VSQG waste, it rarely exceeded more than 1 percent of the total waste handled. Slightly more than half the facilities provided information on whether industrial waste was accepted. Forty-seven percent of those responding indicated that they handle some industrial waste, and at a few of these facilities industrial waste represents as much as 20 percent of the total waste handled.

Environmental Monitoring

Table II-9 indicates the percentage of MSWLFs that report ground-water, surface-water, air, or gas monitoring. Approximately 38 percent of all facilities monitor ground water; less than half this number monitor surface water; and fewer than 10 percent monitor air emissions or levels of methane gas. Forty-eight percent of the facilities that monitor ground water have one to three upgradient wells and five or fewer downgradient wells, and analyze samples from these wells four times per year. Since the survey, EPA has compiled data on new state programs and now believes that landfills in 24 states (containing 45 percent of landfills) have adequate requirements for ground-water monitoring wells (see Chapter IV for additional information).

Table II-9

Monitoring Performed by MSWLFs

Type of Monitoring	Percentage
Ground water	38.2%
Surface	15.1
Gas	6.4
Air	2.1

Source: EPA Municipal Landfill Survey.

Facility Design

A final characteristic of MSWLFs is the method used for landfilling the waste. Approximately 65 percent of all MSWLFs use a cut-and-fill (or trench) method, which entails excavating an area in which to place the waste and then covering the waste with the material obtained during excavation. The remaining landfills rely on the area-fill method, in which waste is spread uniformly over the surface of the ground and then covered with soil. About 15 percent of the landfills that use the cut-and-fill method in some sections rely on the area-fill method in other parts of the facility. The area-fill method is often used when the landfill site contains a natural depression or slope.

III. Regulatory Options

This chapter describes the five regulatory options analyzed in this RIA for complying with the mandate of HSWA; the options include the final rule, the August 30, 1988, proposal, and three other alternatives. The four regulatory alternatives to the final rule represent regulatory scenarios that vary in their costs and benefits. These alternatives are analyzed in this RIA in order to satisfy the requirements of Executive Order 12291.

The final rule is similar to the 1988 proposal with regard to operating and location criteria; from the standpoint of the RIA results, however, the final rule includes several important changes to the design criteria, ground-water monitoring, and closure and post-closure requirements. These parts of the rule have the greatest effect on the estimates of costs, economic impacts, and health risk and resource damage, so they are the focus of the discussion in this chapter.

The final rule combines elements of a uniform design standard for landfills with a performance-standard approach. The uniform standards are part of the self-implementing aspects of the rule which provide an enforceable minimum standard for MSWLFs. At the same time, the performance standard allows flexibility for MSWLFs in approved states where site-specific factors can be incorporated into the selection of a design for the landfills. The performance standard for designs depends on the ability of a given design to prevent violation of health-based contaminant levels in ground water at an appropriate point of compliance (POC). The rule does not specify the designs that must be used to achieve the performance standard, however, so part of the analysis in the RIA is an assessment of which designs landfills will use in order to meet the performance standard.

The other options, briefly described below, include the August 30, 1988, proposal and three additional alternatives. These four alternatives were analyzed to illustrate how various specifications of the revised criteria would affect the estimates of cost, economic impact, health risk, and resource damage.

- Alternative 1, the 1988 proposal, incorporates performance-based design requirements; the performance standard for new units is linked to a design goal, a health- and environment-based ground-water risk level. States must choose a design goal from within a range of risk levels (10^{-4} to 10^{-7}), and facility owners are then obliged to determine what design would be necessary to meet the goal.

- Alternative 2, the Subtitle C option, is based entirely on a stringent, uniform design standard for new and existing units. The requirements for MSWLFs are similar to those imposed on hazardous waste landfills regulated under Subtitle C of RCRA.
- Alternative 3, the categorical approach, sets design standards for MSWLFs based on the environmental setting in which the units are located. The design requirements are structured to prevent leachate from migrating into the underlying aquifer during the active life of the facility, and the final cover must minimize infiltration into the landfill.
- Alternative 4, the statutory minimum, analyzes an option that incorporates only the minimum requirements mentioned in Section 4010 of HSWA. This option assumes a very narrow interpretation of the statute and relies exclusively on detection and cleanup of releases instead of prevention as well as cleanup.

Final Rule

The final rule includes location and operating standards for MSWLFs as well as specific design criteria, closure and post-closure care criteria, and standards for ground-water monitoring and corrective action. Most of the requirements are applied uniformly to MSWLFs, but the rule incorporates flexibility in setting design criteria and post-closure care periods. The flexibility allowed in the rule is also incorporated into the analysis of the costs, impacts, and health risk and resource damage. Because of the flexibility built into the final rule and because the rule does not prescribe the designs that are used in meeting the performance standard, we have modeled different compliance scenarios that simulate the responses of owners and operators to the final rule. In the next two sections we describe the assignment of landfills to compliance scenarios and the design and operating requirements that accompany those scenarios under the final rule.

Compliance Scenarios

The final rule includes somewhat different criteria for landfills, depending on the adequacy of each state's regulatory program. In states with approved programs that are based on the performance standard, the MSWLF owner or operator has the opportunity to take advantage of the flexibility offered by the performance-based criteria. (Even in approved states, however, the state could decide to implement uniform design standards rather than the performance standard on a site-specific basis.) Landfills in states with unapproved programs will be unable to take advantage of the flexibility provided by the performance standard and must instead comply with the self-implementing provisions of the rule.

To estimate the costs and effectiveness of the final rule, we developed an approach to simulate how the 6,034 MSWLFs would respond to the rule given the range of options available. The first step is to decide which landfills are assigned the uniform designs and which are able to incorporate the flexibility of the performance standard. Once we have made this determination, we must also simulate how the second group of landfills will comply with the performance standard.

We assigned uniform designs to landfills in environmental settings that are likely to pose the greatest health risk and resource damage. As discussed in Chapter IX, landfills in wet climates with short travel times for constituents to migrate from the landfill to the water table have the greatest potential for causing health risk and resource damage. We therefore assigned these landfills the most stringent (the uniform) design. Our landfill design assignment is based on site-specific factors (i.e., potential risk)) and does not distinguish between approved and unapproved states. The assumptions are discussed further in Chapter IV; for now, we may say that we assigned 27 percent of the landfills to this uniform design.

The remaining landfills have some flexibility in meeting the design criteria because they are held to compliance with the health-based performance standard rather than uniform design standards. Because the performance standard is based on human-health measures, we relied on the Subtitle D Risk Model (described in Chapter IV) to estimate the effect of the performance standard on MSWLFs. The model predicts leachate constituent concentrations at a number of hypothetical drinking-water wells downgradient from the landfills. We used these results to identify those landfill settings in which landfills would already comply with the performance standard given their current design, and those that need additional protection against leachate release.

For the 73 percent of landfills we assumed were subject to the performance standard, we used the risk model results to indicate which landfills would meet the MCL-based (based on maximum contaminant levels defined in 40 CFR 264) performance standard. This standard requires that no releases from the unit exceed any MCLs (or background levels if no MCL existed) for any constituents during the active life of the landfill plus the post-closure care period. For landfills where constituents would exceed the relevant standard, more elaborate designs, including synthetic covers, synthetic liners, and leachate collection systems, were added to minimize or eliminate releases. New units may need some or all of these components to meet the performance standard; the designs that we modeled are described in the next section.

Existing landfills are subject to a different performance standard: to minimize the formation of leachate to protect human health and the environment. To simulate the impact of this standard, we assigned different final cover designs to landfills based on their potential for generating leachate.

Finally, to reflect the flexibility in the rule regarding post-closure care periods, we modeled costs for two different post-closure care periods, 10 years and 40 years. The final rule also provides flexibility with regard to other issues (e.g., the point of compliance with the performance standard) that were not captured in this analysis.

Design and Operating Requirements Under the Final Rule

The uniform design standards that apply to landfills in unapproved states are specified in the final rule: these units must have a composite liner consisting of a flexible membrane liner (FML) and at least two feet of recompacted soil with a hydraulic conductivity of no more than 1 x 10^{-7} centimeters per second. The uniform design also requires a leachate collection system (LCS) for all landfills. Finally, because the cover material must be no more permeable than the bottom liner, we also included a synthetic layer as part of the final cover design for these units.

The design components for landfills in approved states are not specified in the rule, but the rule does identify the components most likely to achieve the performance standard. To simulate compliance with the performance standard, we assigned units to one of three designs: the baseline design, which assumes an unlined landfill with a cover consisting of earth and vegetation; an unlined design with a synthetic cover system; and a landfill with a synthetic cover system, a synthetic liner, and a leachate collection system. These three designs were assigned to landfills for modeling purposes only; facilities could use other designs to comply with the performance standard. We describe our modeling assumptions in Chapter IV.

The final rule also includes provisions that have an impact on costs and effectiveness, and that apply uniformly to all landfills (although landfills in approved states have some flexibility in selected requirements such as ground-water protection and corrective action). These additional provisions include:

- Developing procedures for excluding hazardous waste from the landfill
- Monitoring for explosive gases in the subsurface and in structures

- Adding run-on and run-off controls
- Preparing a closure plan
- Providing post-closure of 30 years (with flexibility allowed in approved states)
- Demonstrating financial responsibility for closure, post-closure, and known releases
- Monitoring for ground-water contamination and performing corrective action when required (with flexibility allowed in approved states)
- Restricting the siting and operation of MSWLFs in some settings (with flexibility allowed in approved states)

The analysis of these provisions is described in Chapter IV.

Alternative 1: 1988 Proposal

EPA developed its August 30, 1988, proposed rule based primarily on performance standards for MSWLFs. These standards set objectives for owners and operators in determining how to design and operate new and existing facilities. The proposal provides guidance for how facilities can meet these standards, but includes no specific design requirements. The proposal also includes a set of general facility standards and corrective-action requirements to address releases; all MSWLFs must comply with these provisions.

We modeled the effects of the 1988 proposal at a 10^{-5} design goal (from the allowable range of 10^{-4} to 10^{-7}). The proposal also allows for flexibility in determining the POC for new units to meet the design goal. As a more stringent alternative, the proposal describes the POC at the waste management unit boundary, modeled as a point 10 meters from the unit. States may establish an alternative boundary as the compliance point, but we limited our quantitative analysis in this RIA to the 10-meter POC. Actual effects of the proposal would vary from those estimated depending on the state-selected design goal and POC. We also modeled two post-closure care periods: the 30-year post-closure minimum required by the proposal and a 40-year period. Alternative 1 includes a 30-year minimum during which the facility is maintained and monitored; this first phase of post-closure care is consistent with the care period required under Subtitle C.

As with the final rule, the proposal does not detail the designs required to meet the performance standard, so we simulated the responses of owners and operators to the proposal. In the next two sections we summarize our approach.

Compliance Scenarios

Although the performance standard applies to all MSWLFs under the proposal, and not only to landfills in approved states, and is based on risk and not MCLs, the approach used to model compliance with the proposal is similar to the approach described earlier for the final rule. We again relied on the Subtitle D Risk Model to estimate the effect of the performance standard on MSWLFs, and we used these results to identify the settings in which landfills would already comply with the design goal, given their current design and operation.

The model results under baseline conditions (i.e., current designs) indicated that 61 percent of the MSWLFs would meet the 10^{-5} design goal (i.e., have 300-year average risks below 10^{-5}) if the point of compliance is at the waste management unit boundary. If the design goal were set at a different level (e.g., 10^{-6}), the fraction meeting the goal would change.

MSWLFs that would not comply with the proposed design goal at the appropriate POC must be designed with liners, leachate collection systems, and final cover systems as needed to meet the goal. The specific design needed will depend on the location of the landfill. New units may need some or all of these components to meet the goal; the designs that we modeled are described in the next section.

Existing units are subject to a different performance standard (to prevent infiltration after closure), which can be met by changing the cover design. Retrofitting liners and LCSs onto existing units was rejected by EPA as impractical, possibly environmentally hazardous, and disruptive of ongoing solid-waste management activity.

Requirements Under the Proposal

The 1988 proposal, like the final rule, does not explicitly define the design requirements, but to simulate compliance with the design goal, we assigned units to the same three designs used in the final rule for compliance with the performance standard: an unlined landfill with a cover consisting of earth and vegetation; an unlined design with a synthetic cover system; and a landfill with a synthetic cover system, a synthetic liner, and an LCS. Again, these three designs were assigned to landfills for modeling purposes only; other designs could achieve the performance standard.

Like the final rule, the proposal includes other provisions for new and existing units:

- Developing procedures for excluding hazardous waste from the landfill
- Monitoring for explosive gases in the subsurface and in structures
- Adding run-on and run-off controls
- Preparing a closure plan
- Providing for two-phased post-closure care
- Demonstrating financial responsibility for closure, post-closure, and known releases
- Monitoring for ground-water contamination; establishing trigger levels and a ground-water protection standard (GWPS); and performing corrective action when required
- Restricting the siting and operation of MSWLFs in some settings

Further descriptions of how each of these provisions is analyzed are contained in Chapter IV.

Alternative 2: Subtitle C Design Standards

This option is more stringent than either the final rule or the proposal. First, it specifies that all units must comply with a uniform set of design requirements; meeting the performance standard, for example, will not excuse a landfill from cover and containment system requirements. General facility standard requirements similar to those in the final rule are also imposed on landfills under this option. These include post-closure care (analyzed given a range of 10 to 40 years), corrective action where required, gas monitoring, run-on and run-off controls, a plan for excluding hazardous waste, and a closure plan.

This alternative assumes that a backup cover system is always necessary to ensure that leachate generation is minimized. To this end, the alternative calls for installation of a composite cover system at closure. The system consists of a synthetic layer plus a layer of recompacted clay. Both new and existing units must use this cover system.

New units must include a redundant containment system similar to that specified for hazardous waste landfills under Subtitle C of RCRA. The system includes two synthetic layers, with an LCS above each synthetic, and a supporting clay layer to delay leachate migration if leachate escapes the two synthetic layers. As with the cover system, this containment system is more stringent than any system modeled under the final rule.

Finally, the ground-water sampling and testing procedures for new and existing units are similar to those specified for the detection monitoring under Subtitle C. The detection monitoring program described in 40 CFR 264 is used instead of the final rule program in Part 258.54.

Alternative 3: Categorical Design Standards

Alternative 3 represents a categorical approach for determining the necessary cover and containment system designs. The categories are based on location characteristics that, in turn, define a set of design requirements. The emphasis of this alternative is on meeting engineering requirements instead of health-based requirements.

The underlying principles for this alternative are that the design must prevent liquids from reaching the aquifer during the facility's active life, and that the final cover must minimize the infiltration of liquid into the waste. The designs we selected to meet these standards include various combinations of containment systems, leachate collection systems, and final cover systems. The assignment of landfills to specific designs is described in Chapter IV.

In addition to these location-based design standards, this alternative includes the same general facility standards described above for the final rule. These include post-closure care (analyzed given a range of 10 to 40 years), ground-water monitoring, and corrective action as required.

Alternative 4: Statutory Minimum

Alternative 4 imposes on MSWLFs only ground-water monitoring, corrective action as necessary, and post-closure monitoring (for both 10-year and 40-year care periods). No additional design components such as covers or liners are added, and no general facility standards are included (e.g., location standards, run-on and run-off controls);

both new and existing units are required to monitor ground water at the unit boundary. Compared to the other regulatory options, this alternative is the most similar to the statutory minimum (i.e., ground-water monitoring, location standards, corrective action as necessary) mandated under HSWA.

The modeling assumptions for all the regulatory options are discussed in detail in Chapter IV.

IV. Methodology: Cost Analysis

This chapter describes the methodology we used to simulate the costs of the final rule and the four alternative approaches. This chapter also presents an overview of our analysis of changes to the waste management system that are under way, such as regionalization (closing small landfills and consolidating waste in fewer, larger units) and expanded use of combustion and recycling as waste management options. Because these changes are occurring independent of the Subtitle D regulatory changes, we have analyzed the extent to which they lower the costs of these options.

This chapter begins with a general discussion of the cost analysis, which estimates the incremental costs of the regulatory options using a baseline of current practices.

The first section describes the cost model in detail, focusing first on the physical design aspects and then on the cost estimates. Then we describe model inputs to demonstrate how the model simulates the effects of the various options analyzed. Finally, we discuss adjustments to the model's cost results so that we can express the costs of different options in a consistent way (e.g., incremental costs per ton of waste) and so that the results are consistent with the needs of the economic impact analysis and the comparative cost and benefit analysis.

The second section describes the application of the model to the various regulatory options presented in Chapter III, covering the baseline specifications and the specifications for the final rule and the four regulatory alternatives.

The third section describes how we scaled the model results up to national cost estimates. Results of the cost analysis are presented in Chapter VII.

The final part of the chapter presents an analysis of regionalization, recycling, and combustion as factors that may lower the costs of compliance.

To develop a methodology for the cost analysis, we considered both the need to estimate incremental costs and the need to integrate the cost analysis with the benefits analysis, which simulates the performance of new landfills under baseline conditions and under each regulatory option. The basis of the cost analysis is a facility design and cost model that takes selected user-supplied variables (such as tons per day and any of hundreds of design specifications) and produces the specifications and costs for a landfill.

Design and Cost Model

The design and cost model first identifies (or allows the user to specify) the various steps involved in designing, constructing, operating, and closing a landfill; combines this information with data on costs and fees for the components; and estimates the total cost of the landfill. The cost components include a wide range of items, such as land, clearing, excavation, equipment, labor, liner materials, and environmental-monitoring devices.

Facility Design

The model's initial routine calculates the physical dimensions and variables of the landfill. The model includes several basic assumptions (e.g., the landfill unit is square) but allows the user to specify a range of variables:

- Waste characteristics: quantity, density, and operating life
- Unit dimensions: depth of excavation, below-grade slopes, cover slopes, and height above grade
- Containment and cover designs: drainage systems, cover materials and thicknesses (up to ten layers), containment system materials and thicknesses (up to ten layers), leachate collection and treatment systems, and gas migration controls
- Monitoring options: gas monitoring probes, continuous gas monitors, ground-water monitoring, surface-water monitoring

The model then calculates the materials and activities required to support these options:

- Labor inputs: hours needed to design, construct, and operate systems at the facility
- Materials costs: costs for land, heavy machinery and equipment, roads, fencing, buildings, fill material, and liner and cover material
- Indirect costs: engineering, testing, quality assurance (QA), contingency, and contractors' fees on materials and installation

Within a model run, the model specifies the components required to construct and operate a landfill and the costs associated with each. The physical design parameters are also used as inputs to the Failure/Release Submodel of the Subtitle D Risk Model, described in Chapter VI.

Facility Costs

The model summarizes the estimated facility costs for each design simulated, measuring costs for direct capital, indirect capital, operation and maintenance, closure, and post-closure. Corrective action costs may apply to some facilities, but these are estimated by the risk model (because the need for corrective action is simulated by the risk model) and are added separately. The corrective action cost methodology is described later in this chapter.

Direct Capital

Capital costs are substantial for these landfills; they include the large initial costs, such as equipment purchase, land purchase, excavation, site preparation, design, and any environmental monitoring or containment systems. Capital costs also occur intermittently throughout the operating life of the landfill when "packages" of equipment used to operate the landfill are purchased. The pieces of equipment used depend on the size of the landfill. The model specifies 13 different equipment packages, depending on unit size. At small landfills, the appropriate package includes multipurpose machines like bulldozers that can serve several functions, including clearing, moving waste around the site, and compacting waste. Larger landfills can afford more specialized, efficient equipment at a lower cost per ton of waste managed. Capital costs for covering the landfill occur at closure and are discussed below.

Indirect Capital

Indirect capital costs constitute a substantial fraction of the total cost of capital:

- Contractor's fee: profit for construction contractor
- Construction and field expenses: cost of temporary construction and other overhead
- Spare parts: cost of equipment and parts for maintenance
- Contingency: costs for unpredicted events and design changes
- Inspection and testing: cost for a testing company to ensure that design specifications are met
- Quality assurance: additional fee to cover added cost of inspecting, testing, and documenting that design specifications for containment and cover systems have been met

The percentages used for each of these costs is set by the user.

Operation and Maintenance

The dominant component of annual operation and maintenance (O&M) at a MSWLF is labor costs. The model determines the need for labor hours on the basis of the quantity of waste handled; the different categories of labor hours required are linked to the equipment costs described earlier as direct capital costs. Other annual cost components include equipment maintenance and fuel, fees for licenses or permits, environmental-monitoring costs (if any), and costs for leachate management (if applicable). For this analysis, all collected leachate is assumed to be trucked off site to a publicly owned treatment work (POTW). Although other options are available, this is the most common method of leachate management, according to results from the MSWLF facility survey.

Closure

The dominant cost at closure is for the final cover applied to the landfill. The model includes the direct and indirect capital cost components for the cover, its design, the appropriate drainage system, and final vegetation for the site.

Post-Closure

All the regulatory options call for post-closure care after the unit no longer accepts waste. The costs of this care may include applicable environmental monitoring, leachate treatment, cover and slope maintenance, and annual inspections.

Generic Inputs to the Model

We used the model to analyze a range of options but, in the interest of simplicity, constrained the number of possible user inputs to focus on design variables affected by the final rule and the alternatives. The specifications affected by each option are discussed later in this chapter. Other physical design variables were set independently of the requirements so they do not vary over the different options. These variables are described below.

Sizes

We selected seven model sizes of landfills to represent the range of sizes observed in the population. The size categories are defined in tons per day of waste. Each landfill is assumed to operate for 260 days per year. Table II-2 lists the size categories and their relative frequency in the surveyed population (see Chapter II for a more detailed discussion). Size is the primary determinant of overall cost; we show later, however, that landfills exhibit increasing returns to scale: the larger the landfill, the lower the cost per ton of waste managed.

Design Inputs

The landfills are assumed to operate in one phase. All the excavation is done initially and the final cover is placed over the waste at the end of the operating life. The side slopes of the excavation are set at 1:1, and the unit depth is held constant at 15 feet; this is consistent with the risk model.

Operating Procedures

All the landfills are assumed to use a cut-and-fill method of operation: the operator excavates a trench and places waste both below grade (in the trench) and above grade (in a mound). All new landfills are assumed to operate for 20 years; existing landfills have remaining lives as reported in the survey.

Adjustments to the Engineering Cost Estimates

The model's unadjusted output presents the sum of all costs (in current, 1986 dollars) for each year of operation and, if applicable, throughout the post-closure period. To apply the results to the economic analysis and take account of the time value of money, the model discounts the costs using a three percent real discount rate to provide the present value of the stream of costs.

Because the emphasis of our analysis is on incremental costs, we used the cost model to generate estimates for a baseline, unregulated landfill and subtracted those costs from the costs for regulated units (the model specifications are described below). The difference equals the incremental cost attributable to the regulations.

As described in Chapter III, several of the regulatory options have different requirements for new and existing landfills. This difference has implications for adjustments to the cost model results, and the approach we have taken is described next.

New MSWLFs

All requirements that affect new units are assumed to apply from the time construction begins. The incremental cost presented is the cost difference between replacing the unit with a new, regulated unit and with a new, unregulated unit, taking into account the time at which construction begins. We have assumed that each MSWLF identified in the survey will be replaced at the end of its active life with a new MSWLF that conforms to the appropriate requirements. If the landfill was selected to close before the effective date, we assumed it would reopen as a new landfill on the effective date.[1]

This assumption provides a reasonable upper-bound estimate of the regulatory costs because it does not allow for lower-cost compliance approaches, such as regionalization, recycling, or use of resource recovery facilities. Later in this chapter we discuss our approach for incorporating these other waste management options into the analysis.

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[1] Because the rule will not take effect until 18 months after promulgation, we assumed that surveyed landfills with five or fewer years of remaining life (due to close in 1986, 1987, 1988, 1989, 1990, or 1991) will not incur any incremental costs for existing units; all will incur incremental costs for new units designed to comply with the new regulation. If, instead, we assumed immediate replacement of these landfills with units that would *not* be subject to the new unit requirements, aggregate costs would be about 6 percent lower.

In analyzing the incremental costs per year, we assumed that the operator will recover the costs over the entire active life of the unit. Given our assumptions, the present value of the incremental cost estimated by the model for 20 years of operation and for the post-closure care period is annualized over the 20-year operating period. A 20-year operating period was selected to be somewhat conservative and yet consistent with the operating lives reported in the landfill survey. Approximately one-third of these MSWLFs have operating lives of 20 years or less. For landfills with longer lives, annualized costs will be lower owing to the increased amortization period for capital costs. The 20-year annualized cost estimate could be used to measure cost effectiveness (e.g., cost per case avoided for new units) and in combination with costs for existing facilities in the economic impact analysis, which is described in detail in Chapter V.

Existing MSWLFs

The approach for estimating costs for existing units is more complicated. As we described for new units, a landfill owner is likely to estimate all the costs that will be incurred over the lifetime of the unit and the revenue required to offset those costs. If EPA promulgates new regulations applicable to the unit, any incremental costs are added to the cost burden for which the owner has already planned. The key point is that between the first year of operation and the year in which new rules take effect, the owner had no notion of the increase and did not set aside funds for such contingencies.

If a landfill has an expected life of 25 years and has 10 years remaining, all the costs paid out over the first 15 years are considered sunk costs and do not affect the calculation of incremental costs resulting from the regulation. The difference that we want to measure is the increase in costs (over the remaining life) brought about by the new regulations. These could include capital costs for new equipment in the year the regulations become effective, incremental O&M costs in subsequent years of operation, incremental closure costs assuming more stringent closure requirements, continued monitoring costs after closure, and, possibly, corrective action.

The incremental present-value costs of such requirements can be annualized over the remaining life of the landfill or some of the cost increases can be passed on to future landfills, particularly when publicly owned. However, the longer the remaining life of the unit, the smaller the incremental cost and the less likely that costs will be passed on to the future. For the economic analysis, we use a comparison between the costs for the existing unit and the combined cost of an existing unit plus a new replacement unit to determine whether or not costs will be deferred. When the annualized cost of compliance at the existing unit is higher than the combined cost of the existing unit plus a new one (i.e., the existing unit has a short remaining life), costs are spread out into the future to mitigate the cost burden. When the opposite is true (long remaining life at the existing facility) and costs at the existing unit are lower than they will be for its replacement, costs are not deferred.

In 24 states and territories (which include 2,702 landfills), EPA found that the requirements for ground-water monitoring wells are similar to those included in each of the five regulatory options including the final rule. In those states and territories, we have assumed that ground-water monitoring wells are already required and no incremental cost for the design, placement, or installation of the wells (or the associated hydrogeologic study) results from the federal requirement.

We also used data collected by EPA that indicate that 18 states (with 3,203 landfills) already have some requirements for containment systems. Under the final rule and the proposal, landfills in these states that are assigned to designs with containment systems receive "credit" for the existing state design requirement. If the facility is assigned a synthetic liner, LCS, and synthetic cover, the only cost attributable to the federal regulation is the synthetic cover.

Finally, landfills in four states (with 454 landfills) have existing containment-system requirements that are equivalent to the uniform standard under the final rule, and final-cover standards that are at least as stringent as the synthetic cover design we simulate to meet the performance standard under the final rule. Landfills in these states incur no additional costs resulting from the federal requirements for containment or cover systems under the final rule, proposal, or Alternative 3. Two of the states also receive full credit for state requirements under Alternative 2 (Subtitle C). The other two receive partial credit for existing state requirements and thus reduce the costs attributable to the federal regulation.

The adjustments for liner and cover requirements were made in the RIA for costs only. The incremental benefits resulting from the regulatory options were not similarly adjusted. The total dollar value of these credits was significant. For the final rule, these credits amounted to more than $450 million (annualized combined costs). For Alternative 2 (Subtitle C), the credits amounted to approximately $150 million.

We may still overestimate incremental compliance cost estimates to the extent that baseline design variables like the vegetative cover are less stringent than those already required in some states. We have also overstated incremental costs because we did not incorporate all state requirements (e.g., run-on and run-off controls, gas monitoring, covers) that may currently exist for MSWLFs. Despite these possible errors, the adjusted baseline is representative of the present situation among the total regulated population of MSWLFs.

Final Rule

We described the elements of the final rule in Chapter III; here we discuss how we modeled the assignment of facilities to designs and how we modeled compliance with the other elements of the final rule.

General Facility Requirements

Regardless of which designs are selected at a particular landfill, all of the units are subject to the general operating standards specified in Part 258:

- Develop procedures for excluding hazardous waste from the landfill
- Monitor for explosive gases in the subsurface and in structures throughout the active life and post-closure care period
- Include run-on and run-off controls
- Prepare a closure plan
- Provide for post-closure care
- Demonstrate financial responsibility for closure, post-closure, and known releases
- Monitor for ground-water contamination and perform corrective action when required
- Avoid some locations in siting new units or close existing units in those settings within five years

All these requirements impose incremental costs above the baseline and, with the exception of financial responsibility, are included in the cost model estimates for all units under the final rule. Hazardous waste exclusion and the closure plan are included as additional labor hours required to complete the tasks. The cost calculations for the other requirements are more complicated.

For gas monitoring, we assumed units would install monitoring probes 500 feet apart on two sides of the landfill's boundary. All landfills have at least four probes that are sampled twice a year. Run-on and run-off controls require construction of drainage ditches around the unit and a collection area for run-off.

Existing landfills in unstable locations must close within five years of the date of promulgation. Of the restricted locations described in the final rule, we analyzed only karst terrane. Survey respondents reporting that their landfill was in a karst area must close within five years, as long as alternatives for waste management are available.

Post-closure care includes maintenance of the cover, continued ground-water and gas monitoring, and leachate collection if appropriate. Because of the flexibility in the final rule, we estimated costs assuming post-closure periods of 10 years and 40 years. The results of both scenarios are presented in Chapter VII.

For landfills in states where wells are not already required, we estimated the cost of a ground-water monitoring system consisting of one upgradient well cluster and a variable number of downgradient well clusters. The clusters include three wells that draw samples from different depths. The number of downgradient well clusters, which is keyed to the length of the downgradient boundary, ranges from four for the 10-TPD unit to 20 for the 1,500-TPD unit. To account for the five-year phase-in of the monitoring requirements, we assumed that the costs for the wells would occur in the first year for 20 percent of the landfills. Each year, another 20 percent would begin monitoring until all were in compliance. Landfills in states where adequate systems are already required will incur no capital costs for wells or for studies to determine well placement.

The cost of sampling for Appendix I constituents is estimated by EPA to be $360 per sample. All landfills incur these sampling costs under the final rule. Each well is sampled twice annually.

Finally, when the risk model indicates that constituents have appeared above trigger levels at the POC, costs are included for assessing and implementing corrective action. Because their incidence depends on landfill performance, these costs are modeled as part of the risk model. The corrective action cost methodology is discussed later in this chapter.

The design requirements differ for new and existing units, and depend on location and whether the MSWLF meets the relevant performance standard. These requirements are described below.

Design Assignments

New Units. As discussed in Chapter III, the final rule institutes a combined approach for determining landfill design for new units: some landfills are subject to a uniform national design standard while others may select a less expensive design that will allow them to meet an MCL-based performance standard.

In order to select those landfills that would need to comply with the uniform standard, we identified landfills in environmental settings that are likely to pose the greatest health risk and resource damage. As discussed in Chapter IX, precipitation and the permeability and thickness of the material beneath the landfill play an important role in determining which landfills will release contaminants to the underlying aquifer. To identify these settings, we used precipitation at the landfill and the travel time.

For precipitation, we assumed that landfills located in areas with more than 40 inches of precipitation per year had an increased risk of producing releases. Precipitation controls the leachate generated in a landfill. Where precipitation is low for most of the year, rainfall may never penetrate the waste to a significant depth, so that most moisture that does enter the unit either evaporates or fills available pore space in the waste. It may take many years for the waste to become saturated and for leachate to begin appearing at the bottom of the unit.

Unsaturated zone travel time is a measure of the time it takes for constituents to travel from the unit to the water table beneath the landfill. The thicker and less permeable the soils beneath the unit, the longer the travel time and the longer it will take for contaminants to get into the ground water. We used a wetting-front equation to compute travel time for 621 landfills in the sample of surveyed MSWLFs; those units where travel time was less than 20 years (the life of new units) were categorized as having a "short" travel time.

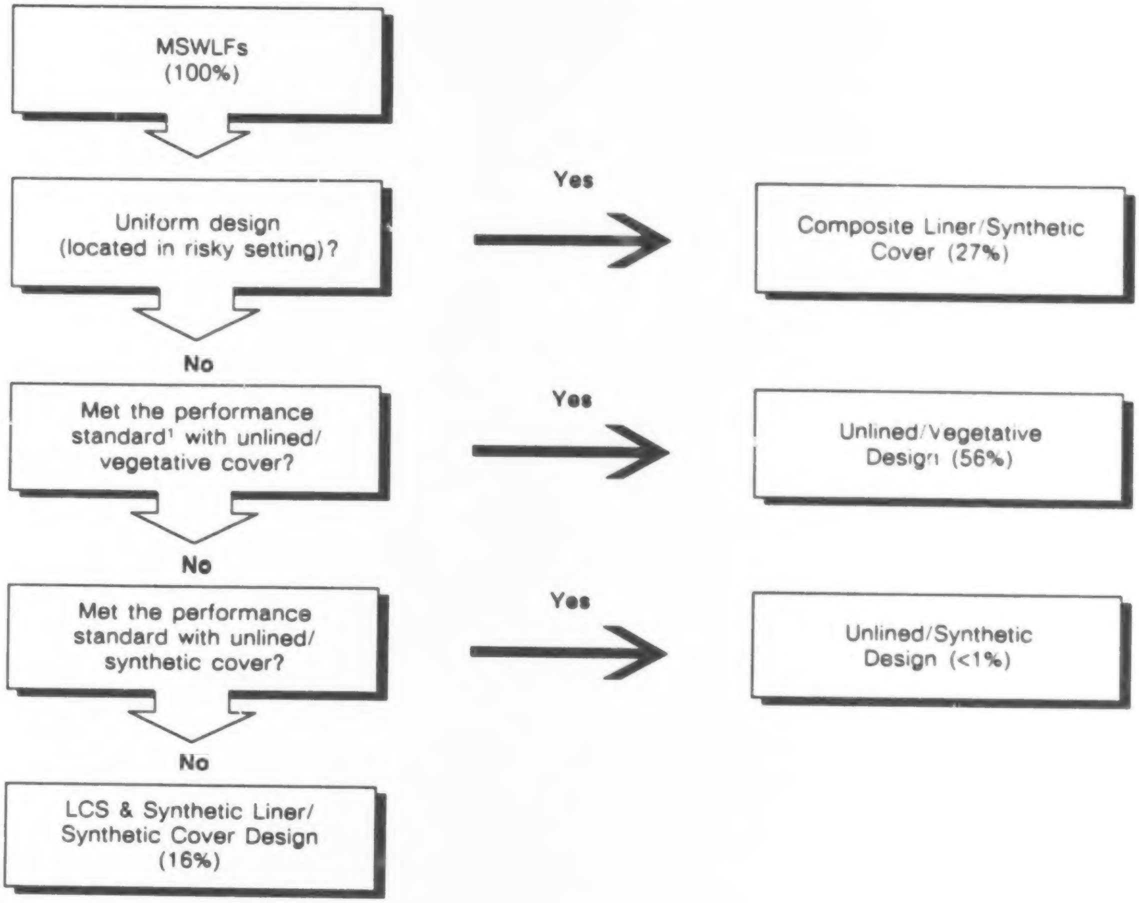
Units with both short travel times and high precipitation have a significant potential for generating releases to ground water and were therefore selected as recipients of the uniform design. We assigned 1,648 landfills (27.3 percent) to the uniform design. As discussed above, because of existing state requirements, landfills in some states will not incur the full cost of this design standard as a result of the final rule; landfills in 22 states are already required to install at least some components of this design.

The rest of the landfills were assigned to designs based on the simulated performance of the landfills. To model the effect of the performance standard on these MSWLFs, we used the Subtitle D Risk Model to estimate constituent concentrations at a compliance point 10 meters from the edge of the landfill. These concentrations were compared to the appropriate standards (MCLs) during the 300-year modeling period; if the standards were not exceeded, we assumed that the landfill would incur no additional costs for design components. If the standards were exceeded, we modeled the unit assuming a synthetic cover to see if the landfill could then comply with the performance standard. If releases from the unit still exceeded the standards, then the unit was assigned a synthetic cover, synthetic liner, and leachate collection system.

Figure IV-1 shows the resulting assignment of landfills to designs under these assumptions. We assigned 27 percent of landfills to the uniform design. The majority of the remaining units would meet the MCL-based performance standard with the baseline design (56 percent of all landfills). Less than 1 percent of the units were assigned to the unlined/synthetic cover design, and the remaining 16 percent require a synthetic liner, leachate collection system, and synthetic cover. It is difficult to predict how landfills will be designed (i.e., whether they use a uniform design or the performance standard), but given our assumptions about potential risk, 27 percent is a reasonable estimate for uniform designs. The designs modeled for landfills under the performance standard are not specifically required by the rule, nor are they the only designs that could meet the standard. They do, however, represent designs that would be appropriate and effective in a wide range of environmental settings.

Figure IV-1

Assignment of Landfills to Design Requirements
Final Rule



[1] The performance standard is based on MCLs for the final rule.

The landfills subject to the performance standard also incur a fixed cost to collect the data necessary to evaluate their ability to meet the performance standard. EPA has determined that this data collection will cost about $15,000 per unit, above the existing costs of landfill design; this cost is borne only by the 73 percent of landfills subject to the performance standard.

The design components are specified in the cost model. The synthetic cover includes 12 inches of fill immediately above the waste, geotextile support fabric, a 30-mil synthetic layer, 12 inches of sand for drainage, geotextile filter fabric, topsoil, and vegetation. This cover system, including design, engineering, and quality-assurance fees, is expected to cost about $27 per square meter, compared with $9 per square meter for the baseline vegetative cover.

The synthetic containment system includes a leachate collection system as well. The design includes (in order) 12 inches of fill directly below the waste, geotextile filter fabric, 12 inches of sand that includes the LCS piping, a 30-mil synthetic layer, and geotextile support fabric. All collected leachate is trucked to an off-site POTW, the most common management option reported in the survey. The cost model assumes use of 30-mil HDPE as the liner material, although the preamble to the final rule recommends use of 80-mil HDPE (which is about 70 percent more expensive). Other 30-mil synthetics such as Driline or Hyperlastic could be substituted at a lower cost than 30-mil HDPE, however, so we believe our estimates of the liner costs are representative of actual experience.

The composite containment system includes all of the same elements of the synthetic liner system except for the addition of two feet of recompacted clay with a permeability of no more than 1×10^{-7} cm/second.

Finally, as discussed above, landfills assigned to the synthetic or composite liner designs in 22 states incur lower costs from the final rule because of existing state requirements.

Existing Units. The final rule includes a different performance standard for existing units. These units must minimize the formation of leachate to protect human health and the environment. In order to model response to this performance standard, we assigned a synthetic cover design to all landfills in settings where net precipitation (the difference between total precipitation and potential evapotranspiration) is greater than 0. Approximately 66 percent of the existing facilities were assigned the synthetic cover system instead of the baseline design.

Alternative 1: 1988 Proposal

Unlike the final rule, the 1988 proposal bases the design requirements entirely on a risk-based performance standard. We relied on the Subtitle D Risk Model to assign landfills to designs based on their ability to meet the performance standard with different design configurations. Before we review those assignments, however, we will discuss the general facility requirements that affect all units under the proposal, regardless of design.

General Facility Requirements

The list of requirements for the proposal is similar to that under the final rule with three exceptions: under the proposal, all owners and operators are required to collect data to allow for an assessment of whether the unit will meet the relevant performance standard, and post-closure care is conducted in two phases. Under the final rule, we determined a unit cost of $15,000 for the data collection, for landfills subject to the performance standard; for this alternative, all landfills will incur this incremental cost. Post-closure care consists of a minimum of 30 years of cover maintenance, ground-water and gas monitoring, and leachate collection if appropriate. After 30 years, the proposal calls for continued monitoring for a period to be determined by the state. To model this requirement, we assumed post-closure care periods of 30 years and 40 years. The results of both scenarios are reported in Chapter VII.

The general facility requirements under the proposal and under the final rule also differ in the number of the ground-water monitoring parameters. The proposal calls for a two-phased monitoring approach that begins with semiannual monitoring for 15 water-quality parameters and over 50 specific constituents. The cost per sample for this first phase is estimated at $644. When these parameters indicate a change in ground-water quality, the second phase begins; this includes quarterly sampling for contaminants that have increased above background. If these contaminants then exceed the ground-water protection standard, corrective action is triggered. The RIA models the sampling costs at $644 until corrective action is triggered. As a result, we may slightly overestimate monitoring costs since the phase-two monitoring under the proposal would likely include fewer constituents.

The remainder of the general requirements are the same as for the final rule and are modeled in the same way in the cost analysis.

Requirements Based on the Performance Standards

In addition to the general facility standards presented above, certain design features may be necessary so that the MSWLFs can meet the relevant performance standard. As discussed in Chapter III, the proposal does not explicitly require any of these landfill designs, but it does describe that the cover, containment, and leachate collection systems will have major effects on ground-water quality.

To model costs and effectiveness, we assigned landfills to specific designs, based on the simulated performance of landfills. As under the final rule, we evaluated compliance with the performance standard using the Subtitle D Risk Model.

We assumed, for purposes of this analysis only, that all landfills would be required to meet a 10^{-5} design goal (from the allowable range of 10^{-4} to 10^{-7}). The landfills with risks below 10^{-5} under baseline conditions (unlined with a vegetative cover) were excluded from any further design requirements and from corrective action, because risks at these units would never exceed the 10^{-5} threshold. Those with higher risks were assigned to increasingly stringent designs, as discussed below.

New Units: Design Assignments. New units are subject to all the general facility standards and may also be subject to design standards to ensure compliance with the design goal. To assign landfills to designs, we estimated the risks from landfills at a 10-meter compliance point under increasingly stringent designs. Those that did not meet the design goal of 10^{-5} under baseline conditions were modeled assuming a synthetic cover system. If the unit still did not meet the design goal, we estimated risk assuming a synthetic cover system plus a synthetic containment system with leachate collection. Landfills that still could not meet the performance standard were targeted for corrective action.

Figure IV-2 shows that at the 10-meter compliance point, 61 percent of the MSWLFs met the 10^{-5} design goal with an unlined unit and a vegetative cover. We then estimated risks for the remaining units with a synthetic cover; 11 percent of the total met the design goal with the cover system, but 28 percent did not. This final group was assigned to a design with synthetic cover, containment, and leachate collection systems.

The specific designs we modeled are not specifically required by the proposal, but they are likely to be effective in a wide range of settings. The design components are the same as those described for the final rule.

Finally, some states already have regulatory requirements for liners and leachate collection systems. To "credit" the cost estimates for landfills in these states, we adjusted the incremental costs for landfills that (1) were shown in the analysis to need a synthetic liner and leachate collection system and (2) were in one of the 22 states that EPA selected as having roughly equivalent liner and leachate collection system requirements.

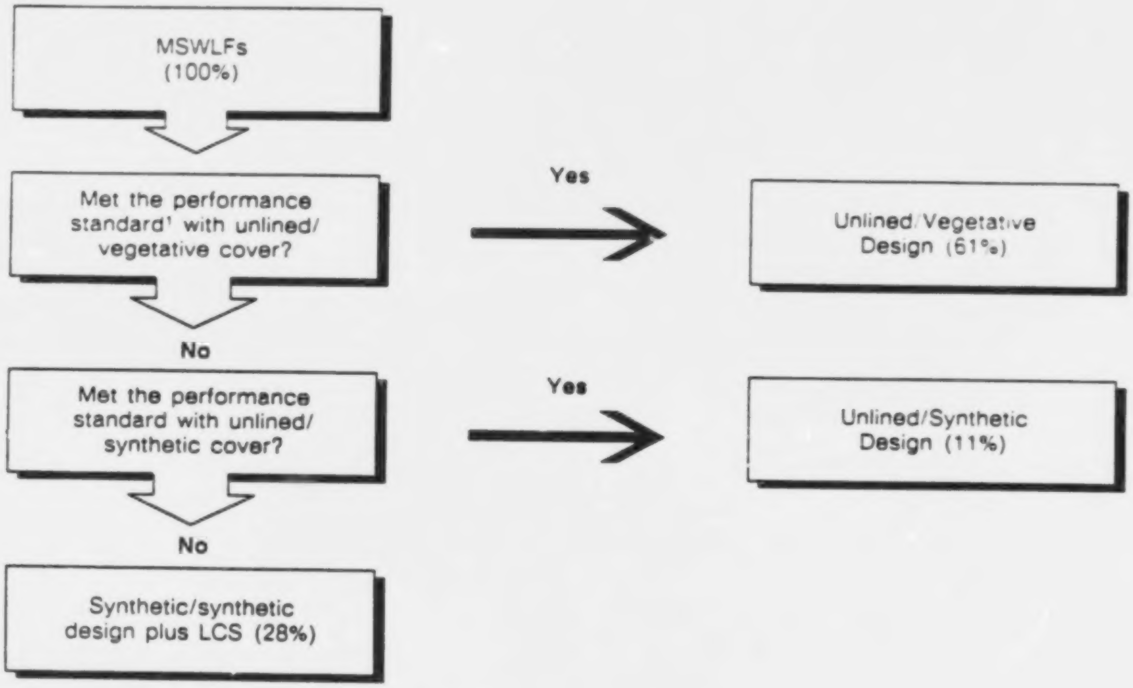
Existing Units. Under the proposal, existing units must be designed to prevent infiltration. As a result, the only design option we modeled was the use of alternative cover systems. The cover was the same described above for new units and was assigned according to net precipitation (as for the final rule).

Alternative 2: Subtitle C Containment and Cover Systems

Under this alternative, uniform standards are imposed on all landfills, so no costs are included to determine compliance with a performance standard. The alternative is more stringent than the final rule with respect to containment systems, cover systems, and ground-water monitoring parameters. Existing unit requirements are the same as for new units except that the containment system and leachate management system requirements do not apply.

Figure IV-2

Assignment of Landfills to Design Requirements
Proposal: 10-Meter Well


MSWLFs
(100%)
Met the performance standard[1] with unlined/ vegetative cover?
Yes
Unlined/Vegetative Design (61%)
No
Met the performance standard with unlined/ synthetic cover?
Yes
Unlined/Synthetic Design (11%)
No
Synthetic/synthetic design plus LCS (28%)

[1] The performance standard is risk-based for the proposal.

The system of ground-water monitoring wells is assumed to be the same as for the final rule, and landfills in states with current monitoring programs will incur no incremental capital costs. The sampling parameters, however, are different, because they are consistent with parameters tested under detection monitoring in Subtitle C. The modeled cost per sample increases from $360 under the final rule to $522.50.

The cover placed over new and existing units will comply with current RCRA Subtitle C requirements described in 40 CFR 264.310. We modeled this cover as a composite system consisting of two feet of recompacted clay, geotextile support fabric, 30-mil synthetic material, 12 inches of sand, geotextile filter fabric, and six inches of topsoil. The difference between this design and the synthetic cover used in the final rule is the addition of a clay layer. The change adds about 33 percent to the total cover cost if the clay is available on-site. If clay is brought from off-site, the cost of the composite cover is almost twice that of the synthetic cover. On-site clay was assumed to be available when clay or sandy clay was reported as the primary or secondary soil type in the 1986 survey.

Finally, the containment system design for new units is similar to that specified under Subtitle C in Section 264.301. It consists of a composite liner and a top liner designed to provide primary leachate collection. We specified that the system would include three feet of clay, a 30-mil synthetic layer, 12 inches of sand for a secondary leachate collection and detection system, another 30-mil synthetic layer, 12 inches of sand to include the primary leachate collection system, geotextile filter fabric, and 12 inches of topsoil. This containment system is two to three times more expensive than the synthetic system described for some landfills under the final rule. Because the costs depend on the on-site or off-site availability of clay, we estimated costs both ways. As with covers, we assumed that clay was brought from off-site unless the landfill owner or operator reported in the survey that clay or sandy clay was available at the site. Landfills in four states (454 units) already face state requirements for at least some components of the liner and cover design; in the cost analysis, landfills in those states incur lower costs as a result of the federal regulation.

Finally, to reflect the flexibility granted to states, we estimated costs assuming that the post-closure care period could vary between 10 and 40 years.

Alternative 3: Categorical Rule Based on Location

This option focuses on engineering requirements due to location only, and does not consider health-risk or resource-damage criteria. Under this alternative, landfills are assigned to designs based on the local climate and hydrogeologic factors that control the potential for releases.

The general facility standards described for the final rule are also added for the categorical alternative. Post-closure care is modeled both at 10 years and at 40 years for all landfills. Monitoring is included for all landfills, but capital costs are subtracted for those in states with existing ground-water monitoring requirements. That monitoring is phased in over five years and the costs per sample are the same as under the final rule.

In addition to the general facility standards, this option includes a range of design options, assigned to landfills based on site-specific information on the potential for releases to the ground water. We used information from EPA's 1986 survey of MSWLFs to assign designs to 621 landfills based on climate and hydrogeology. The results were then extrapolated to the population of 6,034 active MSWLFs.

Existing Units

The only design option we modeled for existing units was the use of a synthetic cover system at closure. This is the same system assigned to some units under the final rule and was assigned in our analysis to landfills based on annual net precipitation. As we did for the final rule, we decided that units with positive net precipitation (about 66 percent) have a greater potential for infiltration, and therefore are assigned this cover. All other landfills have the same vegetative cover that is used in the baseline, except that the general facility standards increase the quality assurance fees.

New Units

New units are assigned to one of seven different design categories, developed for the purpose of modeling costs and effectiveness. These designs were developed based on two principles:

- The design must prevent liquids from reaching the aquifer during the facility's active life.
- The final cover must minimize leachate generation by minimizing the infiltration of liquid into the waste.

The designs we developed are not necessarily the only ones that would meet the performance standards. Given the site-specific nature of this alternative, however, we were compelled to make assumptions about the designs to model this alternative's cost and effectiveness. We will describe how we assigned MSWLFs to the different designs and then summarize the number of landfills per design.

Cover Systems. The assignment of covers was the same as for existing units: MSWLFs with positive net precipitation (about two-thirds) were assigned synthetic covers; all other units had a vegetative cover.

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Cover Systems. The assignment of covers was the same as for existing units: MSWLFs with positive net precipitation (about two-thirds) were assigned synthetic covers; all other units had a vegetative cover.

Leachate Collection Systems. To prevent the release of leachate to the aquifer during the active life of the unit, this regulatory approach requires an LCS in areas of high annual precipitation (greater than 40 inches per year) and in areas with low precipitation if at least one of three conditions is met:

- More than one foot of leachate could accumulate at the bottom of the unit.
- The unit is located within the saturated zone.
- The unit has a flexible membrane liner.

Any of these circumstances could result in a sufficient volume of leachate generation to threaten release to the aquifer during the operating life. Across the 621 landfills for which adequate data were available, we estimated that 75 percent would need an LCS to meet the standards: 34 percent are in areas of high precipitation, and 41 percent meet at least one of the three conditions listed above.

Containment Systems. Our assignment of containment systems to MSWLFs was based on the requirement that they prevent releases to the aquifer during their active lives and on the relative costs of various systems. Figure IV-3 shows the assignment of new landfills to the LCS and liner designs described for new units under the categorical alternative. First, we will review liner assignments for units that also have an LCS.

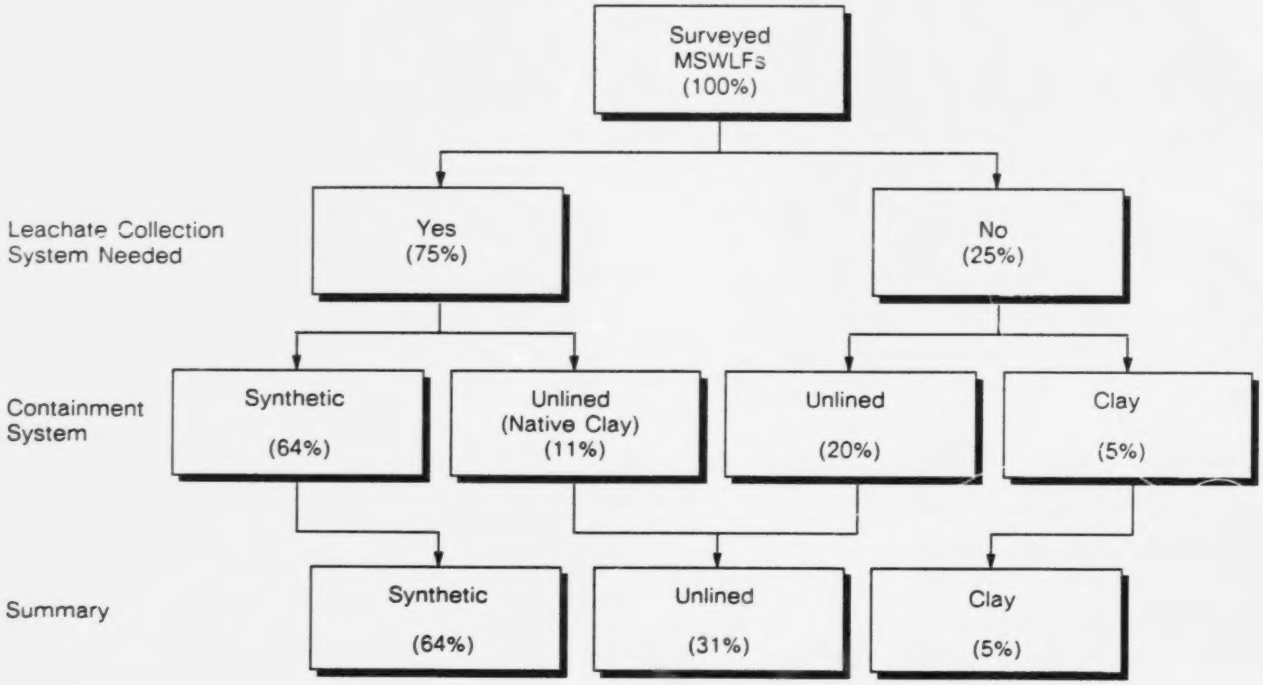
MSWLFs that are assigned an LCS may need to provide an impermeable layer to ensure that the LCS will efficiently remove leachate. If an LCS is installed over permeable material, the system will not be capable of removing an adequate volume of leachate before it permeates the soil. Of the 75 percent of landfills with an LCS, we estimated that 64 percent have soils that are so permeable that a synthetic liner would be required to achieve high LCS efficiency. The remaining 11 percent of the landfills are located over clay and do not need a synthetic layer as part of the LCS. In these cases the LCS is installed directly on carefully graded native soil.

For MSWLFs without leachate collection systems, we assigned containment system designs to keep leachate from reaching the underlying aquifer during the operating life. In these cases, we selected containment systems to delay the transport (increase the time of travel) of leachate from the unit. In most of the 25 percent of landfills without leachate collection systems, the native soils are sufficiently thick or have sufficiently low permeability to provide adequate delay for leachate that might leave the unit. Twenty percent of the landfills therefore require no containment system. The remaining 5 percent could achieve the necessary delay most efficiently by installing a clay liner; we determined that two feet of recompacted clay would provide adequate delay to meet the criterion of no releases to the aquifer during the active life of the landfill.

Figure IV-3 summarizes these LCS and containment system assignments. Sixty-four percent of the MSWLFs had a leachate collection system and a synthetic liner. Eleven percent had an LCS but were unlined, and 20 percent had neither an LCS nor a liner. Finally, 5 percent of the landfills had no LCS, but had a two-foot clay liner to delay the travel of leachate from the unit to the aquifer.

Figure IV-3

Assignment of Surveyed Landfills to Containment System Requirements[1]
Alternative 3



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[1] Extrapolated from 621 surveyed landfills.

Summary of Designs. Table IV-1 summarizes the various combinations of designs that we assigned to landfills. The synthetic/synthetic design with LCS is the most common design, assigned to 56 percent of the landfills. We should emphasize that these assignments are not meant to define the only possible interpretation of the categorical alternative: they represent our best effort at specifying designs to meet the criteria presented earlier. This assignment was necessary to estimate the costs, impacts, and effectiveness of the alternative.

Four states (containing 454 units) were assumed to already require sufficient containment and cover designs to meet these requirements. Landfills in these states did not incur incremental costs for liners and covers as a result of this regulatory option.

Table IV-1

Summary of Designs for Alternative 3

LCS	Containment	Cover	Percentage[1] of MSWLFs
No	Unlined	Vegetative	18%
No	Unlined	Synthetic	2
No	Clay	Vegetative	5
Yes	Synthetic	Vegetative	8
Yes	Synthetic	Synthetic	56
Yes	Unlined	Vegetative	4
Yes	Unlined	Synthetic	8

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[1] Does not add to 100 percent because of rounding.

Alternative 4: Statutory Minimum

This alternative applies uniform criteria to new and existing landfills, but the requirements are much less extensive than for any other option. The only difference between this alternative and the baseline is that we included incremental costs for ground-water monitoring during the active life and post-closure care period as well as corrective action costs as necessary. The capital costs for monitoring and well-placement studies were waived for landfills in states identified as having monitoring requirements already. We assumed that monitoring would be phased in over five years and that the sampling costs are the same as those modeled for the final rule, \$360 per sample. Finally, post-closure care was modeled to last for 10 years and for 40 years; the results of both scenarios are reported in Chapter VII.

Costs Dependent on Risk Model Results

The Subtitle D Risk Model estimates the releases and transport of contaminants through the unsaturated and saturated zones. We used the corrective action component of the risk model to determine the need for corrective action and the costs of corrective action. The corrective action cost submodel provided cost estimates for the design, construction, operation, and maintenance of corrective actions in response to releases from landfills. For this analysis, the corrective action technology (CAT) modeled for these units was limited to recovery wells; costs to replace water supplies in lieu of plume cleanup were estimated elsewhere in the model and were used along with cleanup costs to compute total corrective action costs at existing units.

Our general approach for modeling corrective action was to assume that plumes from new landfills would be completely cleaned up while plumes from existing landfills would be addressed more flexibly. To include in the modeling some recognition of the flexibility available for corrective action, we modeled corrective action for existing units under two approaches based on plume size. The first was an active plume cleanup similar to the approach modeled for new units. The second approach relied on replacing ground-water supplies instead of immediately cleaning up the plume. In Chapter VII we report the fraction of MSWLFs assigned to each of the approaches. The methodology for estimating replacement costs is reported in Chapter VI under the discussion of the resource damage component of the Subtitle D Risk Model. It is difficult to capture accurately the range of flexible responses to the regulatory options; our assumptions were simply estimates made for modeling purposes. Our assumptions about corrective action modeling are described in detail in Chapter VI.

Corrective Action Costs

The risk model estimates the total present value of costs for the corrective action based on the size of the plume and the duration of the action. The methodology used for computing corrective action costs is briefly described below.

Recovery well systems are designed to remove contaminated ground water, treat it, and then return the water to the aquifer or discharge it to surface water. The systems are installed above the plume and pump ground water from the entire plume. We assumed that the recovery wells begin operating one year after corrective action is triggered. The length of time the wells are operated depends on the mobility of the constituents in the ground water, the flow rate of the aquifer, and the rate at which contaminated ground water is removed.

Recovery well costs include direct capital costs, indirect capital costs, and O&M costs. The first step in computing the costs of recovery wells is to calculate the dimensions of the floating plume in the year it is detected. The plume discharge rate is calculated based on flow field variables (porosity, velocity, and thickness, as described later in Chapter VI). The pumping rate, pumping duration, number of wells necessary, and well depths are then computed as a function of the plume discharge rate. We calculated direct capital costs for wells and effluent treatment plants based on well dimensions and the pumping rate. Indirect and O&M costs were estimated as a percentage of direct costs. As with other costs, these were incurred over time and discounted to year zero of the simulation period.

Choosing a Corrective Action

After the model calculated the cost and effectiveness of each corrective action, we selected as the CAT of choice the lowest-cost action that reduces contaminant concentrations below the target level at the 10-meter monitoring well. In previous model runs, recovery wells were found to be the least-cost CAT for virtually all plumes. Consistent with this finding, recovery wells were selected as the chosen CAT for this analysis.

Assigning Corrective Action Costs to Landfills

The Subtitle D Risk Model provides the outputs used to assign corrective action costs to landfills. The model analyzes a range of typical environmental settings, defined by climate and hydrogeologic factors discussed in Chapter VI, and it identifies the landfill sizes and designs in these settings that are likely to trigger corrective action (i.e., where constituents exceed trigger levels). We calculated from the model results the fraction of landfills in each setting that will trigger corrective action and the weighted average cost they will incur. The costs were a function of the unit size and design, CAT, net infiltration, depth to ground water, and ground-water flow rate. Finally, we assigned these costs to individual landfills based on survey information that indicates those most likely to have high risks (e.g., those with short times of travel for constituents to nearby wells). In Chapter VII we present the results of the analysis including the incidence and costs of corrective action under each option.

Assignment of Costs to Actual Landfills

The cost analysis produced a results matrix of year-by-year capital and operating costs for each design option by each of the seven model landfill sizes, which ranged from 10 to 1,500 TPD. Our next step was to use these cost estimates to project actual compliance costs for the 621 active landfills in the survey. These landfills provided sufficient information on their environmental setting to allow us to assign designs to specific landfills for all of the regulatory options. To assign costs, we developed a cost function for each design option that related incremental compliance cost per ton of waste to landfill size. With this function we could then enter the actual size of an active landfill and directly estimate the incremental cost of a specific option for that landfill. For options like the final rule that include several unique landfill designs, separate cost functions were developed for each design so that each landfill could be costed according to its assigned requirements. Separate cost functions were also developed for new and existing facilities.

The incremental cost for new facilities was calculated based on a 20-year operating life. For existing facilities, the remaining life of the landfill as reported in the survey was employed to estimate compliance costs.

Key assumptions in assigning costs to the 621 active landfills were as follows:

- Real discount rate is 3 percent.
- Remaining life at existing landfills is as reported in the survey, but with these exceptions:
  - Remaining life is limited to 50 years for those landfills where the data are reported.
  - When not reported, the remaining life is assumed to be eight years (the median remaining life as reported in the landfill survey).
- New MSWLFs have a 20-year life.
- Costs for existing landfills are deferred beyond the closure date when the levelized cost across the existing landfill plus a new landfill is lower than full cost recovery at the existing landfill.
- Waste landfilled is assumed to be 0.68 tons per person per year.
- Landfills operate 260 days per year.

Developing National Cost Estimates

Once the incremental compliance costs for each of the surveyed landfills were calculated, they were scaled or weighted up to national cost estimates. The weighing factors (developed as part of the Office of Solid Waste's 1986 Survey) differ because the sample for the survey was stratified based on landfill size. The sample included a higher proportion of large facilities (greater than 500 TPD) than small ones. Therefore, the weighing factor for small landfills is greater than the one for large units. These factors relate the number of the landfills surveyed to the current universe of landfills. The weights are shown in Table IV-2.

Table IV-2

Derivation of Weights to Compute National Costs

	Below 500 TPD	Greater than or Equal to 500 TPD	Total
Universe	5,784	250	6,034
Survey	552	69	621
Weight (universe/survey)	10.48	3.62	

This waiting approach means that in developing the total cost for all MSWLFs, we multiply each cost by 10.48 for landfills below 500 TPD, and by 3.62 for all landfills above 500 TPD.

Cost Adjustments Based on Changes in Waste Management Approaches

The analysis of costs and effectiveness in the RIA generally assumes that current waste disposal practices remain unchanged over time, even after the regulation becomes effective. For example, the cost analysis assumes that when they reach their closure date, landfills close and are immediately replaced with a new unit of the same size. We will continue to report our results under these assumptions to provide an upper bound on costs and impacts of the rule.

In reality, however, the solid waste management system is dynamic, reflecting shifting landfill size and location, recycling, waste combustion, and even source reduction. We have devised a sensitivity analysis approach for factoring these changes directly into our estimates. First, we have quantitatively analyzed regionalization (shifting waste from small landfills to larger, regional facilities). Second, we conducted a sensitivity analysis using waste discard estimates to test the possible impact on compliance costs resulting from projected waste reduction, recycling, and waste combustion. In this section, we will discuss our approach for adjusting costs based on the trend toward regionalization, recycling, combustion, and source reduction. In Chapter VII, the national cost estimates we present will reflect both the upper cost estimate (no additional regionalization, combustion, recycling, or source reduction) and a second, lower estimate.

Background

The trend toward larger, regional MSWLFs has been quite evident over the past decade. In 1977, *Waste Age* magazine[2] reported that 18,539 municipal waste disposal units were operating in the United States as of 1974; by 1986 that number had dropped to just over 6,000 units. Analysis of data from the MSWLF survey indicates that the typical size of landfills has increased since the 1960s. The data show that almost 80 percent of all MSWLFs opened prior to 1966 were smaller than 50 TPD; only 60 percent of landfills opened between 1982 and 1986 were that small.

There are two major reasons for regionalization. The first is economic: with the number of landfills declining, increased demand for disposal and greater awareness of landfill design and operating standards have driven up disposal costs. *Waste Age* magazine reported a 23 percent increase in tipping fees between 1987 and 1988. These increases suggest that it is becoming more economical to haul wastes longer distances to take advantage of the economies of scale than to operate small landfills close to generators. The second reason for regionalization is the political difficulty of siting new facilities. The rate of new landfills built slowed considerably during the 1980s, according to survey results, and this trend shows no sign of reversing. This encourages regionalization because the effort to site a large regional landfill is not much greater than that required for a small landfill, leading to additional scale economies.

Given the regionalization trend, the cost of compliance with the regulatory options may actually decline in the future; over time, increased amounts of waste will flow to larger facilities, which have lower unit compliance costs (i.e., lower costs per ton of waste disposed) than smaller landfills.

The second set of issues we considered was shifts away from landfilling resulting from source reduction, recycling, and municipal waste combustion (MWC). As waste shifts away from landfills, the tons of waste disposed and the costs of compliance with the regulatory options will both decline. Our analysis examines EPA projections for MSW discards between 1986 and 1995 and adjusts compliance costs accordingly.

Because of the uncertainty inherent in projections about waste generation, recycling, source reduction, regionalization, and combustion capacity, we propose this sensitivity analysis as one possible scenario. For our results, however, we present a range of cost estimates to reflect this uncertainty (see Chapter VII).

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[2] *Waste Age*, January 1977, p. 26.

decision rules we used to predict the responses of owners to closure of their existing landfills.

The table shows that all landfills smaller than 125 TPD (the upper bound of the 75-TPD category) will eventually shift to larger units. The economic analysis suggests that the smallest landfill size that would operate economically is the 175 TPD model size.

Table IV-3

Projected Transfer of Wastes to Regional Landfills

Existing Landfill Size Category (tons per day)	Population Density Categories (people per square mile)				
	<17	17-40	40-90	90-235	>235
10	750T	1,500T	375	375	375
25	750T	1,500T	375	375	375
75	750T	1,500T	1,500T	375	375
175		1,500T	1,500T	375	750
375					
750					
1,500					

T = transfer station built as part of the decision to shift.

Impact of Regionalization on Compliance Costs

The second component of the regionalization analysis applies these decision rules to estimate the quantity of waste that will shift to larger landfills and the resulting savings in regulatory compliance costs. We computed the quantity of waste that would shift from one size category to another based on the decision rules in Table IV-3 and the quantity of waste landfilled in units that were scheduled to close as of the effective date of the rule and 5, 10, and 20 years later. If we measure total disposal in tons per day, the survey indicates that just over 800,000 TPD is currently disposed (see Table II-3). The landfills that were scheduled to close before the end of 1991 *and* would shift to larger units accounted for 68,249 TPD of capacity, or nearly 9 percent of the total. Table IV-4 summarizes the share of the total waste quantity affected by these two criteria over the next 20 years. The table shows, by the size of the old landfill, the share of the total tons per day of capacity that is going out of service and will shift to larger units. After 20 years, about 21 percent of the capacity will have shifted to larger units.

Table IV-4

Cumulative Percentage of Waste Shifted to Larger Landfills[1]

Old Landfill Size (TPD)	Percentage of Waste Shifted As Of:			
	Effective Date	**Five Years**	**Ten Years**	**Twenty Years**
10	30.5%	49.3%	74.0%	85.3%
25	35.5	51.0	67.2	83.1
75	33.1	50.0	69.3	87.8
175	35.2	47.8	63.4	79.7
375	0.0	0.0	0.0	0.0
750	0.0	0.0	0.0	0.0
1,500	0.0	0.0	0.0	0.0
Totals	8.5%	12.2%	16.6%	20.8%

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[1] Reflects decision rules in Table IV-3 and reported closure dates in the MSWLF survey.

While these shifts will save communities both baseline costs and regulatory compliance costs, for our analysis of regulatory cost adjustments, we included only the savings resulting from lower compliance costs per ton; the remaining savings in baseline disposal costs would occur even without the regulation and should not be factored into the estimates of offsets to the compliance costs. The landfill owners are not able to realize these compliance cost savings without some offset, however; those landfills that shift to larger units as a result of the regulatory cost savings will also incur incremental transportation costs as a result of the regulation. These costs are offset against compliance cost savings for landfills that shift due to the regulation. Those that shift as a result of baseline cost savings will incur no additional transportation cost from the regulation.

In order to estimate savings, we computed an average compliance cost per ton for each of the size categories in the analysis. This average masks differences between new and existing units as well as different regulatory compliance strategies (e.g., different design standards under the final rule), but is an appropriate measure for an aggregate analysis of costs. The incremental transportation costs incurred by those landfills shifting as a result of the regulatory costs are offset against these regulatory cost savings. In Chapter VII we present the total cost adjustments to the national cost estimates for each regulatory option. Again, because of the uncertainty associated with these estimates, we have incorporated these assumptions into the lower estimate in a range of compliance cost estimates presented in Chapter VII.

Analysis of Future Municipal Waste Discards

We assumed that shifts away from municipal waste landfills would reduce the total cost of compliance with the regulatory options because these shifts reduce the total tons of MSW that must be disposed in landfills. These shifts are occurring even without the added influence of the regulations, so we assumed that the baseline shifts simply reduce the total tons of MSW that will incur incremental compliance costs. We assumed that a uniform percentage of waste would shift away from all landfills, regardless of size or their decisions to regionalize.

To estimate the impacts on costs, we analyzed projections of future waste discards prepared for EPA. According to EPA's *Agenda for Action*, 131 million tons of MSW were discarded in landfills in 1986. Current estimates by Franklin Associates[4] place 1995 discards at 106 million tons. These estimates were derived using consistent methodologies and are therefore comparable. Since the decline in discards amounts to a 2 percent annual rate, we used the rate of decline as an initial proxy for the impact of additional source reduction, recycling, and combustion.

This 2 percent rate is likely to overstate the reduction somewhat, because residuals from combustion and recycling must also be landfilled and are *not* included in these estimates. Since these residuals as well as commercial components of the waste stream are likely to increase, we chose to analyze the results assuming a decline in total MSW landfilled of only 1 percent each year over a 20-year period. After 20 years, the total quantity of waste entering landfills will decline by approximately 18 percent, given these assumptions. We assume that this decrease will translate into an equal decrease in the total costs of compliance (after regionalization is taken into account).

As with the regionalization analysis, this sensitivity analysis includes uncertainty and so we have incorporated both analyses into our estimates of the lower bound of costs in Chapter VII. The upper end of the range assumes that waste generation, waste combustion, recycling, and the distribution of landfills across size categories remain constant.

Summary

This chapter described the approach for estimating compliance costs for the five regulatory options, including the final rule. The cost methodology relied on a design and cost model that computes the physical and design specifications for MSWLFs as well as the costs to design and operate the landfills. We relied on the Subtitle D Risk Model and data from the MSWLF survey to assign landfills to designs and compliance scenarios under each of the regulatory options, and then computed national cost estimates. Because of the dynamic nature of the solid waste market, we have incorporated into our final cost estimates two adjustments to evaluate the sensitivity of the results to increased source reduction, recycling, and combustion and the trend toward larger, regional landfills. The cost results are presented as a range to reflect this uncertainty.

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[4] Franklin Associates, Ltd., *Characterization of Municipal Solid Waste, 1990 Update*, prepared for U.S. EPA, June 1990.

V. Methodology: Economic Impacts

Overview of Approach

The economic impact analysis assesses the impacts of alternative regulatory approaches on those who will face increased costs and analyzes how those impacts will be distributed.

As Chapter II indicated, over 80 percent of all landfills that handle municipal solid waste are owned by local governments, including counties, cities, towns, and villages. The governments that own these landfills will bear the initial impacts of new Subtitle D regulations. These governments will then pass the increased costs on to their own citizens and to neighboring governments that might also use the landfill. A government may pass on higher costs in the form of increased taxes or fees or decreased services of other types if the community is constrained by its budget. Ultimately, the households that use the landfill to dispose of their waste will most likely pay the increased costs of landfill operation.

About 15 percent of all MSWLFs are owned by private operators. The solid waste handled by these landfills comes from governments or households that contract for the landfills' services. Because all landfills will experience higher costs as a result of new regulations, most private operators will be able to pass these higher costs on to the governments and households that they serve. Private landfills that have much higher regulatory costs than average may have some difficulty passing these costs on to their customers if they are located near resource recovery facilities or other landfills that could serve the same communities at a lower cost. When costs cannot be passed on, the private operator will incur an economic impact that could force the landfill to close. For the economic analysis, we assume that landfills will pass through costs and that the resulting economic impacts will accrue to municipalities and households. While this assumption is valid in the majority of cases, we supplement the economic analysis with a review of cost impacts on the 15 percent of landfills that are privately owned.

Our approach for conducting the economic impact analysis is divided into two steps. In the first step we assess the ability of municipalities[1] to pay for increased landfill costs, since these entities are most likely to face higher solid waste disposal costs in the short term. In the second step we review the ultimate impact of new regulations on the citizens of municipalities, since they will pay for higher landfill costs, either immediately or over a longer period as taxes are raised or other services reduced. These two analyses are conducted sequentially to determine the overall economic impact on households from regulatory options affecting municipal solid waste disposal landfills.

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[1] The terms "municipalities" and "communities" are used in this report to identify local governments responsible for solid waste disposal. These governments include counties, cities, towns, and villages. Many states and the U.S. Department of the Census use the term "municipality" to refer only to incorporated jurisdictions that satisfy certain conditions.

The analysis of economic impacts on municipalities is subdivided into two elements. The first element of the municipal impact analysis involves assessing the impact of regulatory costs on these municipalities by calculating annual costs as a percentage of total municipal expenditures and comparing this ratio with a predetermined threshold level.

The second element is an assessment of the baseline financial capability of the community. Unlike the measure of costs as a percentage of expenditures, which addresses the magnitude of costs, the financial capability element considers communities' ability to absorb regulatory costs (i.e., their practicable capability). For simplicity, this practicable capability measure can be thought of as a composite score that characterizes economic vitality (e.g., population growth, level of poverty), debt burden (e.g., debt per capita), and financial management (e.g., operating surplus/deficit, expense levels) of a community. The score categorizes communities' financial capabilities as weak, average, or strong. To perform these analyses, we compiled a detailed database on the financial and demographic characteristics of local governments, as described below.

This approach for conducting community impact analyses was selected because the overall financial capability score of a community will not change significantly with the imposition of costs: increased operating expenditures for MSW management will not change many of the indicators used to measure financial capability. In addition, it would not be appropriate to presuppose the reaction of a community to higher landfill costs. Some communities will increase taxes or fees, while many others will reallocate available funds to meet higher landfill costs. In the area of debt impact, it is also not clear how a given project will be financed. Many communities will use pay-as-you-go financing as they always have, others will take on debt, and the remainder will turn to private contractors who will raise their own capital.

The second half of the impact analysis focuses on the ultimate ability of a municipality's citizens to pay for regulatory costs and assesses the absolute and relative impact of regulatory costs on households. The absolute impact is measured in terms of total cost per household, and the relative impact is measured using costs as a percentage of median household income. First we describe the economic impact measures computed for communities and individuals. Second we describe the development of the database and measure of communities' financial capability. The results of the economic impact analysis are presented in Chapter VIII.

Measures of Economic Impact

To evaluate the economic impact of additional regulatory costs, we examined several approaches that measure economic impact, including evaluating increased costs relative to communities' and households' wealth, income, and economic potential. Such analyses are commonly used by other analysts and have been used by EPA in other municipal programs.

For example, EPA's Construction Grants Program has used a number of measures to identify potentially high-cost projects. These measures include:

- Total annual cost per household and as a percentage of median household income
- Capital cost per household
- Total annual cost as a percentage of total personal income
- Total and general obligation debt as a percentage of fair market value of property
- Outstanding debt per capita

Some of these measures, such as the debt measures, are more meaningful when assessing impacts on a single governmental entity (as in the case of the Construction Grants Program), because the analyst can better determine the extent to which the community will actually rely on debt financing.

In our analysis we examined three measures of impact:

- Total annualized cost as a percentage of expenditures (CPE)
- Total annualized cost per household (CPH)
- Total annualized cost per household as a percentage of median household income (CPMHI)

The first measures impact on municipalities, while the second and third focus on impacts on individual households.

Cost as a percentage of expenditures provides a clear indication of how expensive the regulations will be relative to communities' current operating budgets. Most communities, except those already financially weak, can accommodate small increases in expenditures with relatively little disruption. Even strong communities, however, may have difficulty adjusting their budgets to accommodate large increases in expenditures, especially in the short term. We address in detail communities' ability to cover these costs in a later section.

Communities typically spend less than 1 percent of their budgets on solid waste disposal. Calculations using data from EPA's Municipal Landfill Survey, for example, indicate that municipal solid waste disposal costs average approximately 0.5 percent of communities' total expenditures. In comparison with other municipal services, costs at this level represent a very small obligation. Table V-1 shows the average cost for other selected services as a percentage of total community expenditures in 1982, according to the Census Department.

Table V-1

Costs for Community Services

Service	**Average Percentage of Expenditures**
Education	36%
Police	5
Sewage disposal	3
Fire protection	2
Solid waste collection and disposal	1

On the basis of these data, we established a severe impacts threshold at 1 percent of total community expenditures. If incremental costs of compliance for a community exceed 1 percent of total expenditures, we describe the impact as severe. While an increase in costs equal to 1 percent of expenditures would be a large percentage increase for disposal costs, the level of spending on other services suggests that even our "severe" impacts are relatively minor in absolute terms.

We can compare this threshold to other impact measures as a way to test its reasonableness. The Construction Grants program did not identify a parallel impact measure because costs were always evaluated in terms of effects on households. (The program was required to recover costs based on user fees, so a comparison to community expenditures was not relevant.) Economic impact assessments performed for private entities often include two similar thresholds, however: cost greater than 1 percent of sales and annual cost increases of greater than 5 percent.[2] The 1 percent CPE threshold is analogous to the cost as a percentage of sales measure, assuming that local revenues are like sales for a private sector firm. Increasing local expenditures by 5 percent would represent a tenfold increase above the current expenditure level of 0.5 percent on disposal and would be much too high as a threshold for significant impacts.

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[2] Assessment of Economic Impact Methodologies and Industry Economic Data Sources, prepared for Economic Analysis Branch, Office of Solid Waste, by Temple, Barker & Sloane, Inc., June 1985.

Our current threshold of 1 percent is consistent with the "percentage of sales" threshold and is appropriate given the relatively low level of current expenditures devoted to waste disposal. Communities exceeding the 1 percent CPE threshold will face at least a doubling of current MSW disposal costs.

Cost per household and cost per household as a percentage of MHI measure the impacts of the regulations on the people who will ultimately pay these costs and who actually generate the majority of municipal solid waste (as Chapter II indicated, the majority of the waste stream at MSWLFs consists of household waste).

Two threshold levels are used for cost per household: one that provides a comparison to households' current expenditures on MSW and one based on MHI. Estimates from the MSWLF survey indicate that households in most communities spend between $10 and $30 per year on MSW disposal, excluding collection and transportation costs. These costs may be understated somewhat because they are reported annual expenditures and may not include amortization of capital equipment and land. We will discuss these estimates further in Chapter VII. A threshold of $220 per household was used to measure severe impacts. This threshold was calculated as 1 percent of the MHI of all the communities in the analysis. The 1 percent threshold has been used by the Office of Municipal Pollution Control for evaluating Construction Grants programs (see below). An incremental regulatory cost of $100 per household per year was selected as an intermediate threshold for impacts, primarily for reporting purposes. The more important threshold is the $220 level. The $220 threshold represents a significant increase above the reported $10 to $30 annual CPH computed from EPA survey data. As described in Chapter VII, we believe these estimates to be understated because communities often do not know or do not report the true cost of disposal. In order to provide an absolute cut-off, we relied on the 1 percent threshold established for cost as a percentage of MHI for Construction Grants. We recognize, however, that as MHI varies, the CPH threshold should also vary. For that reason, we also computed CPMHI as a more precise measure of community-specific household impacts. Although this cost represents a large percentage increase in many households' disposal costs, it represents a relatively small absolute charge.

EPA's Construction Grants Program established threshold levels for CPMHI. Their criteria ranged from 1.0 percent of MHI for low-income communities to 1.75 percent of MHI for high-income communities.[3] A threshold of 1.0 percent has been adopted in our analysis as a significant level of burden.

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[3] U.S. Environmental Protection Agency, Office of Water Programs Operations, *Financial Capability Guidebook*, Washington, D.C.: (February 1983).

Estimating Compliance Costs for Municipalities

To calculate the measures of economic impact discussed above, we first had to determine the proportion of landfill compliance costs that would be passed on to municipalities and their citizens. To achieve this, we analyzed the subset of municipalities that are served by the 621 surveyed landfills for which we had prepared compliance cost estimates. The municipalities participating in these landfills were identified in several different ways. First, the survey form requested a listing of jurisdictions served. This information was useful, but often nonspecific or incomplete. Responses ranged from specific municipalities to "portions" of towns or counties. The municipalities listed in the survey that were also included in the comprehensive municipal database were identified as valid landfill participants. If a municipality could not be located in the municipal database, it was not included in the economic analysis, since we had no direct way of obtaining its demographic and financial information.

We utilized several information sources to identify additional municipalities that are probably served by the landfills in the survey, but were not listed in the survey responses. First, we compared survey responses to our previous assignments of municipalities to all landfills using a complete listing of landfills from the EPA State Census of landfills, which contains 7,645 landfills. This assignment of municipalities to all the landfills was done with a distance-minimizing algorithm that used the latitude and longitude of municipalities and the landfills. The distance algorithm assigned municipalities to the closest landfill, while imposing some restrictions on which landfills could serve which municipalities (e.g., limiting crossing state lines). The resulting landfill assignments were used to supplement the MSW landfill survey data.

In the many cases where the survey respondents indicated that the landfill served the "entire county," we were able to assign municipalities located within those counties to that landfill. In cases where no information was provided on the municipality served, the city where the landfill is located was at least assigned as one of the municipalities served.

The next step was to ensure that we did not assign too many municipalities to the landfills, either through our additional assignments or through assigning the entire population of a large city to a given landfill, while only a portion of that population is actually served by that landfill.

We used population and waste quantity information as a check against this potential problem. The sum of the populations of each town assigned to a landfill was compared with the reported population served by that landfill. When population assigned exceeded the reported population served, adjustments were made by eliminating municipalities that had been added using the supplement assignments. When the population served by the landfill was not provided in the survey, we utilized the waste quantity received information to test the reasonableness of the population we had assigned to the landfill. We calculated waste generated by this population using the estimate of 0.68 tons of waste per person per year and compared it with the waste quantity reported in the survey. When these figures seemed unreasonable we checked reported waste quantity and our estimated waste quantity generated against reported landfill capacity. These checks, which were based on population served, waste quantity generated, and landfill size, were used successfully to ensure that municipalities were properly assigned to landfills.

The links between landfill compliance cost estimates and the participating municipalities then served as the basis for cost estimates used in the economic analysis. The total number of municipalities identified as using the surveyed landfills was 1,810. Thus, just as the surveyed landfills provide a sizeable sample of all MSWLFs, the links between landfills and municipalities identified a similar sample of municipalities for the impact analysis. The municipalities identified did not cover all the 621 landfills analyzed, but did cover 532 (86 percent) of them.

Once the subset of municipalities was determined, the incremental costs at the landfills were apportioned from the landfills to the municipalities. This was done on a unit cost basis (dollars per ton) to account for imperfect matches between a landfill and the exact number and size of the jurisdictions served and for the potential overlap of landfill services with landfills that were not included in the survey. The apportioned compliance cost estimates were then used to measure economic impact on the municipalities and their citizens.

The results from the impact analysis are weighted up to national totals for all 29,017 municipalities. The weights were based on the population size and financial condition (described in the next section) of the municipalities included in the sample and were calculated by dividing the total number of municipalities in a size and financial condition category by the corresponding number of municipalities in the impact analysis. The final weights applied to each category are presented in Table V-2.

Table V-2

Scaling Weights Used to Extrapolate from the Municipal Sample to all Municipalities

	Financial Condition		
Population Size	**Weak**	**Average**	**Strong**
Less than 500	22.8	22.4	18.4
500-1,000	17.1	18.0	14.2
1,000-2,500	20.3	16.0	16.4
2,500-5,000	13.9	12.8	17.0
5,000-10,000	10.7	15.6	15.6
10,000-25,000	9.4	13.3	14.3
25,000-50,000	11.5	13.7	10.7
50,000-100,000	7.7	13.9	7.5
Greater than 100,000	5.8	6.6	11.0

Source: TBS calculations.

Database Description

Developing a database on financial and demographic characteristics of municipalities was the first step in conducting the financial capability analysis. This section, which describes the database, is divided into four parts: sources of data, number and type of jurisdictions included, complexities encountered, and financial and demographic information.

Sources of Data

As we began to build the database of financial and demographic characteristics, we examined a number of commercially available data sources. These are sold by private vendors, which include CAC International, Inc.; Moody's Municipal Research Department; and Donnelly's Marketing Service. They all repackage Census Department data for various marketing purposes. We found that they were generally incomplete for this analysis. This led us to choose the Census Department, which is the original source of all the commercially packaged data, as the data source for our analysis.

The two primary data sources for the database are the *1982 Census of Governments* and the *1983 City and County Data Book*. Complete government censuses are conducted every five years, so the *1982 Census of Governments* data are the most recent information available on local government finances. Our demographic information comes primarily from the *1983 City and County Data Book*, which was chosen so it would closely match the other census data. We obtained the data from both sources on magnetic tape, matched the different governmental unit coding systems, and merged the information into one large database.

After we compiled the initial database, we talked extensively with the Chief of the Census Department's Government Finance Branch, to ensure that our interpretation of the data was correct. In addition, we spoke with another expert on census information at Syracuse University.

Number and Type of Jurisdictions Included

The complete database includes selected information on revenues, expenditures, debt, and demographic characteristics for 29,017 governments. The 29,017 governments were chosen after we examined every state individually and determined the main providers of local government services in each.

The primary governments in the database are in four categories:

- County
- Municipality
- Township
- Rest of county

We use the "rest of county" category as an accounting tool in counties that are dominated by one or more municipalities but that have a significant population not contained in one of these incorporated areas. A rest-of-county entry exists wherever 20 percent or more of the population, or more than 5,000 people, do not live in incorporated areas. The full county generally is not the primary provider of local government services, which is why so few counties are included in our database of primary local governments (Table V-3).

Table V-3

Breakdown of Primary Government Entities

	Number	Percentage of Total
County	197	0.7%
Municipality	17,618	60.7
Township	8,780	30.3
Rest of county	2,422	8.3
Total	29,017	100.0%

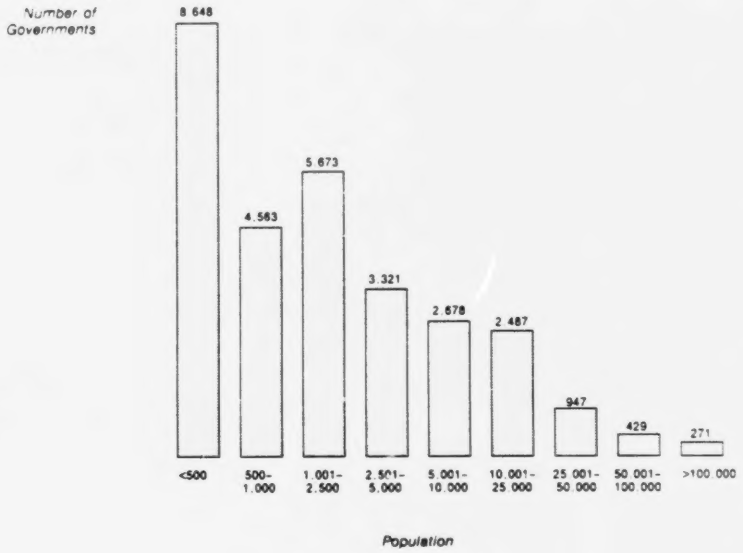
The number and type of governments in the database vary greatly by state. For example, Washington, D.C., is counted as a single municipality, whereas Pennsylvania contains 2,564 townships and municipalities. The New England states tend to have many municipalities and towns, since the states are composed almost entirely of incorporated areas. Conversely, the southern states have stronger county governments, and much of the population lives outside the boundaries of incorporated areas.

Figure V-1 illustrates the size characteristics of the 29,000 governments of interest. The chart identifies the number of governments by a variety of population size categories. It illustrates the very large number of small entities, particularly in light of the fact that the regulatory flexibility analysis guidelines suggest a population cutoff of 50,000 in defining small governments. More than 75 percent of these jurisdictions have fewer than 5,000 residents.

Figure V-1

Number of Governments Providing General Government Services

By Population Categories



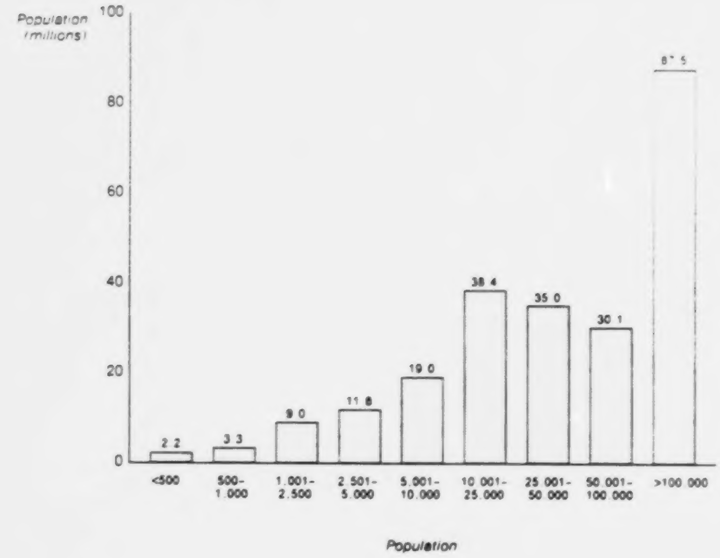
Source: TBS calculations.

Figure V-2 presents the number of people that live in each government size category. Here it becomes apparent that a substantial portion of the U.S. population lives in a few large metropolitan areas. Conversely, the thousands of small towns represent a relatively small fraction of the total U.S. population; the 76 percent of jurisdictions with populations of less than 5,000 account for only 11 percent of the population.

Figure V-2

Total Population of Communities Represented by Each Government

By Population Categories



Source: TBS calculations, based on 1980 Census

Complexities Encountered in Building the Database

We faced two major obstacles in creating the database: allocating county and special-district financial data to smaller governments, and coping with the wide variation in governmental organizations found across the country. The first problem occurred because counties (and special districts) provide some but not all local government services. Thus, the revenue, expenditure, and debt statistics for the individual municipalities do not accurately describe the financial situation of their residents, since the residents are also responsible for a share of the county's, and in some cases special district's, revenues, expenditures, and debt. We solved this fundamental problem by allocating county and special-district financial data to smaller governments on the basis of population, a process we call "allocating down."

When these figures are combined with those already recorded for municipalities, they provide a more accurate financial characterization of a municipality's residents.

The problem of variation in governmental organization has many forms. Although organizational forms and terminology tend to be fairly consistent in a given region, this is not always the case. In disparate areas of the country, the inconsistencies are more marked.

These problems are most evident in the classification of towns and townships and in states where counties rather than municipalities are the major suppliers of local government services. For example, Indiana has township governments covering all its area and population. Naturally, these overlap with many other governmental units in the state, and debt is allocated to Indiana's population through several overlapping levels of government. Indiana, however, is the only state where townships cover the entire population. In five other states, operating townships comprise all areas not served by municipalities. Maine would also fall into this category, but it contains "unorganized territory," which lacks any local government. In ten other states, some municipalities operate in territories also served by townships.

In some instances, townships share exact geographic borders with the overlapping governmental unit. In these cases, the names and populations of the two governments are often identical, and financial information can be allocated equally to all persons in the region. This greatly simplifies the process of assessing a community's financial health. Frequently, however, the overlapping government units do not exactly coincide. In these cases, the allocation of debt and revenue and the financial capability assessment are substantially more difficult to perform.

Several states are characterized by weak municipal and strong county governments. Municipal revenue, debt, and expenditure figures in these states do not accurately reflect the true financial situation since the municipalities receive substantial services from the county, the costs of which are not reflected in the municipal data.

Five states or groups of states required substantial data manipulation to determine accurate financial and demographic data for primary local governments. The first group consists of seven states, primarily in the Midwest, that contain overlapping jurisdictions. The townships in these states overlap the municipalities. In these cases, we took the county financial data, which included township figures, and allocated them down to the municipalities on the basis of population. Our analysis indicated that townships in these states performed such limited functions that this method of allocation should introduce only minor errors.

The second group of states that present difficulties consists of Vermont, New York, and Michigan. These states are similar to the first category, except that villages, not townships, overlap with municipalities. The villages provide only minor services, however, and we solved the problem exactly as we did for the Midwestern states.

In Connecticut, where the township is the predominant form of municipal government, there are 10 boroughs and 3 cities that overlap townships. We manually eliminated these 13 overlapping entities from the database, and added their demographic and financial data to those of the associated township.

The last two categories included states with strong county and weak municipal governments. Maryland and Virginia are two such states, but for these, the solution to the difficulty was relatively straightforward. Because large cities are single county units, we were able to analyze these states by county.

The last "problem" states also have strong county governments that contain significant municipal units within their counties. We encountered this situation in practically every county in a southern state, and in other counties scattered throughout the nation. To solve the problem, we searched the database to find counties in which either more than 5,000 people or more than 20 percent of the total population live in unincorporated areas. In counties that met either of these criteria, we created a "rest of county" entity containing all population not living in incorporated areas. The allocation of county revenue, expenditure, and debt is proportional to population in the affected areas.

Financial and Demographic Information Included

The database contains 36 variables that describe each included government. Eight of these variables were selected to compute the financial capability score; the remaining indicators are purely descriptive and allow us to characterize different segments of the database in a variety of ways. For example, we can use this information to determine whether communities that are adversely affected by more stringent regulations are likely to be rural or smaller communities.

Initially, we examined each of the more than 100 financial and demographic indicators that the Census of Governments requires local governments to record. We needed to narrow this large field to develop a workable database. Since the major component of a bond rating is a financial-capability analysis similar to the one we were undertaking, we began the narrowing process by studying the methods used by the professional bond rating services.

Standard and Poor's (S&P's), Moody's, and Fitch are the three nationally recognized bond rating services. Municipalities issuing bonds frequently ask one of these services to assign a rating to the issue. This rating serves as a measure of the issuer's ability to make payments to the bonds' holders. A bond score is therefore a measure of the financial capability of the rated government. A bond rating also includes a subjective estimate of a community's willingness to pay its bondholders,[4] a judgment not relevant to our analysis. Nonetheless, the methods of professional rating services, if not the final scores, provide valuable insight into the estimation of financial capability.

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[4] "Pitfalls in Issuing Municipal Securities," *Moody's Investors Service*, January 1982, p. 22.

We examined S&P's and Moody's rating methods for General Obligation (G.O.) bonds, which require the issuer to pledge its full faith and credit to bond payments, and which attempt the broadest estimation of financial capability. When rating G.O. bonds, S&P gives the greatest emphasis to:

- Current or total operating deficit
- Outstanding debt
- Analysis of the economic base

The rating service assesses these areas by looking at fiscal factors such as current financial position and fund balance; debt measures, including trends; the community's debt service history; and a number of demographic indicators such as age, education, and wealth of the population. The worksheet included with a bond rating application requests numerous other debt, revenue, and demographic measures. Moody's asks for similar information, information about the community's relationship to the surrounding area, and information about cyclical factors.

Our study of bond raters' methods served as a starting point for our own analysis. We focused on the measures that professional services have found important in determining financial capability. This approach enabled us to trim the large initial list of potential financial and demographic variables to a smaller, more manageable list. Using correlation studies, we found the few indicators that reflect information contained in numerous others. We also examined relationships between the various measures and bond ratings, and tried to confine our field to those widely used indicators that even the smaller governments record.

Our review of the data revealed several important characteristics of local government finances. In the area of debt financing, they often do not issue bonds to finance capital expenditures. We found that approximately half of all jurisdictions have no direct long-term debt. Average direct debt per capita is, therefore, only about $200. Once overlapping jurisdictions such as school districts, special authorities, and counties are considered, however, most local governments are responsible for repaying at least some long-term debt; this overall debt per capita (which includes direct debt) averages about $900. Approximately 40 percent of all long-term debt is used to finance education expenses, although this percentage varies dramatically depending on the size of the government, with smaller entities using much more of their debt to finance education.

One clear pattern in our analysis was the relationship between population size and many of the financial variables. Larger communities tend to have higher debt and expenditures per capita and as a percentage of median household income. The pattern of their spending is also different. Smaller communities issue a much lower percentage of debt to finance utility expenditures such as water, electric, or transit services than do larger communities.

We also examined the relationship between many of the indicators and bond rating, one of the most common summary measures of financial capability. Many of the indicators show a U-shaped distribution with bond rating. For example, communities with the best bond ratings (AAA and AA) have higher-than-average expenditures per capita than communities with the lowest bond ratings (BA and BAl). Because the relationship between many of the indicators and bond rating is not linear, it is difficult to test whether particular indicators are better correlated with bond rating using correlation coefficients or simple regression models.

Our analysis revealed that demographic variables, such as median household income and the civilian unemployment rate, are better predictors of bond rating than many of the debt and revenue indicators. This conclusion is the same as the one reached in a study on a much smaller database reported in the June 1986 issue of the *Government Finance Review*.[5]

Municipal Financial Capability Baseline

The database was used to assess the economic impacts that are likely to accrue to municipalities and their citizens from Subtitle D regulations; we estimated from the database the current or baseline financial capability of 29,017 municipalities. This section describes the objectives of the analysis to assess financial capability, the calculation of a financial capability measure, and efforts to validate the measure.

Objectives of the Financial Capability Analysis

The overall objective of the financial capability analysis is to establish the relative financial condition of municipal jurisdictions around the country. This assessment of financial condition will then be used to determine the economic impact of regulatory costs and the practicable capability of municipalities to meet these costs. In establishing a measure of financial capability, several criteria were employed to structure the analysis. They include:

- Assembling the comprehensive national data set
- Using data describing financial capability from three general areas:
  - Financial management
  - Debt burden
  - Demographics

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[5] See also "Value of Bond Ratings Questioned by a Growing Number of Studies," *Wall Street Journal*, September 17, 1987.

- Producing a summary measure of impact for each jurisdiction
- Measuring financial capability in a relative rather than an absolute sense

Measurement of Financial Capability

To develop a measure of financial capability for municipalities, we examined a variety of financial management, municipal debt, and demographic indicators. We then selected a subset of eight indicators for assessing financial capability. The indicators are:

- Financial management
  - Total taxes as a percentage of median household income
  - Current revenues as a percentage of current expenses
  - Coverage ratio (funds available for debt service payments)
  - Operating expenditures as a percentage of median household income
- Debt
  - Net overall debt per capita (direct and overlapping debt)
- Demographics
  - Median household income
  - Population growth from 1980 to 1990
  - Fraction of the population below poverty level

These eight indicators share several characteristics: (1) they are all widely used as indicators of economic and financial health by others looking at regulatory impacts; (2) they correlate well with other possible indicators that were not included on the final list; and (3) they are important factors in determining a community's bond rating.

To assess the baseline financial condition of municipalities, we developed a composite financial capability measure that characterizes the overall financial health of each municipality. The rating scheme is quintile based: for most indicators, a score of five is assigned when the municipality falls in the top 20 percent of all municipalities, four in the next 20 percent, and so on. For all indicators, a score of five indicates a strong rating, four above average, three average, two below average, and one weak.

Three of the indicators – coverage ratio, population growth, and net overall debt per capita – were handled differently from the others. For the first two indicators, most municipalities were simply rated as average, with a few categorized as either weak or strong based on the value of the indicator for that municipality. In the case of coverage ratio, for example, all municipalities whose coverage ratio exceeded one (approximately 85 percent of the entries) received a score of three (average), while the remaining municipalities were rated as weak. In the case of population growth, governments that lost more than 15 percent of their population and those that grew by more than 80 percent were both rated as weak; the remaining governments (95 percent of all entries) were rated average.

For debt per capita, we assigned communities with per capita debt of less than $518 to the average category; we determined that it was inappropriate to score communities with very low or no debt as strong, because that may simply mean that they are poor risks (i.e., they cannot borrow readily). The remaining 40 percent of the communities received a score of two (below average) or one (weak).

Table V-4 presents the range of values used to assign scores for each of the eight indicators as well as the mean and median values for these indicators.

Table V-4

Indicators Used to Measure Baseline Financial Capability

Indicator	Mean	Median	Strong (5)	Above Average (4)	Average (3)	Below Average (2)	Weak (1)
Taxes as a percentage of MHI	7.3%	6.3%	<4.16	4.16-5.47	5.47-7.09	7.09-9.21	>9.21
Current revenues as a percentage of current expenditures	107%	105%	>114	108-114	103-108	98.4-103	<98.4
Coverage ratio	4	2.5	–	–	>1.00	–	≤1.00
Operating expenditures as a percentage of MHI	18.5%	16.2%	<11.7	11.7-14.5	14.5-18.0	18.0-23.3	>23.3
Net overall debt per capita	$764	$415	–	–	<518	518-837	>837
Median household income (MHI)	$15,209	$14,718	>17,801	15,475-17,801	13,828-15,475	12,073-13,828	<12,073
Population growth (1990 population/ 1980 population)	116%	109%	–	–	85-180	–	≤85 or ≥180
Population below the poverty level	13.4%	12.4%	<8.21	8.21-10.58	10.58-13.14	13.14-17.40	>17.40

Source: TBS calculations based on U.S. Census data.

The composite financial capability score was developed by averaging the scores for all eight indicators. The final determination of financial capability was made by examining the relative value of each government's composite score. The lowest 25 percent of the governments were rated as weak, the middle 50 percent of governments were classified as average, and the top 25 percent of governments were rated as strong.

Validation Efforts

Because the financial capability rating system represents a novel approach to combining census data, we performed validity checks on the data throughout the development of the database.

Data Validation

Developing a measure of municipality-specific financial capability required the compilation of detailed financial data. The generally accepted data required for such an analysis measure indebtedness, fiscal responsibility, and economic vitality for a given jurisdiction. While we considered a number of ways to obtain such data for the analysis, we felt that the most cost-effective and comprehensive approach was to obtain the data directly from the U.S. Census Bureau's general survey and survey of governmental finances.

We identified three types of problems that could lead to data inaccuracies. The first is that information was incorrectly reported to the census at the outset of the census survey. The second problem is that jurisdictions do not use uniform accounting techniques. The third problem is that the Census Bureau may have mishandled some of the data received. All of these potential problems could have occurred for some of the data elements contained in the database.

The task of identifying the magnitude of the first type of error (incorrect reporting to the census) would be enormous, essentially requiring a follow-up census that goes into greater depth than the Census Bureau. Such a validation was not practical or necessary, in our opinion. We believe that the vast majority of the data are accurate. The data were carefully collected by the Census Bureau, an organization that is uniquely qualified to do so, and were used extensively by such firms as Moody's, Inc., for their municipal bond rating service.

The issue of differing accounting techniques is one that we must accept as a given in conducting this analysis. Some jurisdictions maintain their books on a cash basis just as individuals do for federal income taxes. Others use the more widely accepted accrual method, which is used by corporations. The difference between these methods is the timing of recording a cash outlay or inflow. Using a cash basis, transactions are recorded when funds change hands. Under accrual, transactions are recorded when the obligation to pay or receive funds is made. The difference between these accounting methods should be fairly small as the flow of funds into and out of public jurisdictions is relatively uniform; therefore, cash and accrual-basis accounting should produce very similar results. Our conclusion was that different accounting techniques could produce some inconsistencies in the data, but that they would not introduce a great deal of error into the analysis.

Once we were comfortable that the majority of the data would be consistent and relatively error free, we proceeded to analyze the data realizing that our most powerful tool for identifying data irregularities would be the comparison of reported figures across jurisdictions. For all of the indicator variables selected, we carefully followed a procedure of identifying and investigating the extreme outliers in a distribution. For example, as we developed the debt-per-capita variable, we found that Valdez, Alaska, has reported debt of roughly $1 million per person on its books, but it also has the income to cover this debt service. We surmised that a major facility for the Alaska Pipeline was paid for with municipal bonds issued by the city. Without further investigation, we concluded that this may be a legitimate outlier. This type of outlier analysis did not typically attempt to focus on a jurisdiction's unique financial situation, such as Valdez; rather, we reviewed the other financial data for the jurisdiction to establish whether or not the observation was reasonable. We found the data to be reasonable in most cases, but, as expected with such a large data set, a limited number of peculiar entries do exist. Again, the effort required for reconciliation would be enormous and of limited value, given that 99 percent or more of the observations fell within a reasonable range.

The greatest problem that we saw with a limited number of outliers was that they created large standard deviations when calculating means. We were concerned that variables could be overweighed if the absolute magnitude of an outlier were counted in a financial score. To overcome these problems, we decided to use medians, to define the distribution of results in quintiles, and to use a one-to-five scoring system that restricted the range of results.

Overall, we concluded that our data set was far superior to the alternatives. All of the alternatives involved original data collection from a limited set of jurisdictions. Such approaches could improve the quality and understanding of the data in hand, but would still have to be extrapolated to produce national estimates, thus introducing another type of error.

Results Validation

Given a reasonable data set, our next validation issue was whether our use of data to estimate financial capability was correct. This issue was evaluated by analyzing the results of our analysis. Our efforts in this area were limited, although we believe the results are reasonable. Our first exercise in this area was to attempt to correlate bond ratings to our scores, since bond ratings are supposed to be based on many of the same criteria that we used in the financial score. This comparison revealed that communities with low bond ratings generally scored poorly on our system, but those with higher bond ratings did not necessarily score well on our system. This was a puzzling outcome that either meant that our scores were wrong or we did not capture all of the factors considered in setting a bond rating. We reached the conclusion that the latter case is more likely based on similar findings by other investigators (see earlier discussion of bond rating in this chapter) and anecdotal evidence that factors like the quality of bond rating presentation and apparent competence of the local officials play a major role in setting ratings.

The next step in our validation effort was to review qualitatively our scores for 30 Boston-area communities. The results of our scoring were consistent with the overall financial health of these towns.

Waste Shifts and Potential Mitigating Factors

The estimated cost to municipalities from the regulatory options assumes that the municipalities will continue to use their current landfills and then replace these landfills with similar new units once current capacity is exhausted. In reality, some landfills will be replaced by larger, regional landfills to take advantage of economies of scale, particularly in more urbanized areas. Also, the use of waste disposal alternatives such as resource recovery and recycling is becoming more common. In Chapter IV we described our approach for reflecting regionalization, recycling, source reduction, and combustion changes in the cost analysis. We modeled the economic incentives that landfill owners would have to shift from smaller landfills to larger ones, given savings in baseline and incremental costs per ton and offsets in the form of higher transportation costs. We also evaluated the fraction of waste that would shift out of landfills as a result of expanded waste combustion, recycling, and source reduction over the next 20 years. Each of these analyses was conducted at an aggregate level rather than landfill by landfill as the design cost and economic impact analyses were. As a result, while these adjustments provide an indication of how national costs could vary as a result of the final rule, the level of aggregation limits our ability to analyze the accompanying effect on economic impacts.

Overall, we believe that our assumptions about landfill replacement without regionalization overstate costs for many communities. This is of special concern because the majority of the landfills affected by regionalization are small units serving small communities. Ideally, we would use the (lower) adjusted cost estimates to estimate impacts and report impact results over a range. Unfortunately, because of the cost adjustment approach we adopted, we were unable to recalculate impacts that precisely. Given the complexity involved in modeling future behavior across 29,000 primary jurisdictions, the aggregate approach was the only practical one to adopt, but it is critical to emphasize that the economic impact results are overstated somewhat as a result.

One adjustment that we did make to the impact analysis was to project, by landfill size category, how much waste would shift to larger landfills in the future. Chapter VIII includes an assessment of the communities served by these landfills, and the number of communities that would shift from small landfills to larger ones; we assumed that once the shift occurs, the impacts on individual communities would ease because of the much lower compliance costs per ton. Additional factors that would reduce impacts are discussed in the analysis of the results in Chapter VIII.

VI. Methodology: Risk and Resource Damage

Introduction

The Subtitle D criteria revisions must address a diverse population of MSWLFs in a variety of environmental settings. Hundreds of factors affect the nature, extent, and severity of health and environmental impacts from these landfills. We have developed the Subtitle D Risk Model to enable us to evaluate quantitatively the adverse effects of these units on human health and resource value, and to examine the effectiveness of the regulatory options in reducing these impacts. The model couples results from EPA's Solid Waste (Municipal) Landfill Survey with a series of mathematical formulations of engineering, physicochemical, hydrologic, toxicologic, and socioeconomic processes that govern impacts, to provide a framework for evaluating regulatory options.

Our basic approach was to use the Subtitle D Risk Model to simulate risk and resource damage first in the baseline case and then under each of the designs imposed by the regulatory options being considered. For each design, we modeled risk and resource damage in hundreds of scenarios that represent unique combinations of landfill size and design, environmental setting, and exposure distance. We then estimated the frequency with which each scenario occurs in the nationwide population of MSWLFs, and weighted the results for each scenario to reflect its frequency of occurrence.

In this chapter we provide an overview of the Subtitle D Risk Model, and then describe how we used it to analyze the regulatory options.

The Subtitle D Risk Model

We have designed the Subtitle D Risk Model to provide (1) an analytical framework for estimating human health risk reduction and other benefits of regulatory options; (2) a direct link between estimates of regulatory benefits and costs; and (3) scenarios that contain different combinations of design, waste, environment, and response. The model builds directly on the Subtitle C Liner Location Risk and Cost Analysis Model,[1] and has adopted many of its basic characteristics. It is a dynamic model, with time horizons of up to 400 years, although in the model runs made to support this analysis, we simulated 100 years of leachate release and up to 200 years of ground-water transport for each year's release. It models environmental fate and transport and dose-response relationships as deterministic processes, while simulating containment system failure and some hydrologic events as stochastic phenomena. The model focuses on ground water as the environmental medium of concern; releases to air and surface water are not considered. As with the cost model described earlier, some parameters can be varied over a wide range; for others, the user selects from specified "generic" values. Finally, the model resides in an integrated, computerized form that links all the supporting model components and databases.

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[1] U.S. EPA Office of Solid Waste, *The Liner Location Risk and Cost Analysis Model: Phase II*, Draft Report prepared by ICF, Inc. (1986).

The model includes a series of submodels that simulate pollutant release (failure/release and leachate quality submodels), fate and transport (unsaturated zone and saturated zone transport submodels), exposure, impacts (dose-response and resource damage submodels), and corrective action. These are discussed below.

Pollutant Release

The failure/release submodel uses a Monte Carlo simulation model to estimate the probability of failure (defined as release to the unsaturated zone) and the time to failure for landfills, and to estimate the quantity of leachate released. The submodel uses a "fault tree" structure; it traces each possible failure event from all possible combinations of basic events, such as liner failure and the infiltration of liquid, which could combine to cause failure. The model assumes that each of these basic events occurs at random, following specified probability distributions. Using this fault tree structure, the model predicts distributions of the year of failure and the release rates. We used the model to simulate the performance of several combinations of containment and cover systems in eight environmental settings (four annual net infiltration rates and two depths to the water table for each infiltration rate).

Given the stochastic nature of the failure/release component and the deterministic nature of the subsequent transport, exposure, and risk modeling components, we must provide a deterministic link between the failure/release component and the "downstream" submodels. To do this, we generated 250 release profiles using the Monte Carlo simulation and used a statistical clustering technique to select from one to ten typical release profiles. We used each typical profile to estimate impacts, and then weighted the results by the relative frequency with which the profile occurs in the 250 iterations. For this analysis, we selected one typical profile; this minimizes the influence of low-probability, high-consequence outcomes on the model results.

The leachate quality submodel simulates the concentrations of chemical constituents in leachate released from the landfill for up to 200 years (only 100 years of release were modeled in this analysis). To account for differences in the leaching behavior of constituents, the submodel uses three different modeling approaches that simulate the concentrations of three different constituent types. To model inorganics, we developed an algorithm based on the process modeling approach described by Straub and Lynch[2] and the semi-empirical approach described by Wigh;[3] the algorithm calculates constituent concentrations as a function of initial concentration, cumulative infiltration, moisture capacity, an empirical parameter, and an optional constituent generation rate. For phenol (a product of waste biodegradation), we adapted a semi-empirical algorithm described by Wigh[4] that calculates constituent concentration on the basis of parameters similar to those used in the inorganic algorithm. Finally, to model synthetic organic chemicals, we developed a theoretical approach that calculates concentrations on the basis of the partitioning and degradation characteristics of each constituent, initial concentration, and cumulative infiltration. The submodel applies the appropriate algorithm to calculate the concentration of each leachate constituent. The concentration is then multiplied by the annual release volume calculated by the failure/release submodel to calculate the mass flux of each constituent across the landfill/subgrade boundary.

Fate and Transport

Subsurface transport modeling addresses transport through both the unsaturated zone and the saturated zone. The Subtitle D Risk Model uses the McWhorter-Nelson Wetting Front Model to calculate the delay between the time of failure and the time that leachate reaches an underlying aquifer. The mass that breaks through the unsaturated zone then advects and disperses through the ground water. Using an adaptation of the Random-Walk Solute Transport Model developed by Prickett, Naymik, and Londquist, the model simulates ground-water concentrations over a period that covers 100 years of release and as many as 200 years of transport for each year of release. We predict concentrations at seven downgradient distances ranging from 10 meters to 1,500 meters. These distances define the location of wells where exposure might occur. The Subtitle D Risk Model includes 11 ground-water flow scenarios that reflect variations in aquifer configuration, materials, and flow velocity. We assume that the subsurface materials are homogeneous and isotropic.

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[2] W.A. Straub and D.R. Lynch, "Models of Landfill Leaching: Moisture Flow and Inorganic Strength," ASCE, Vol. 108, No. EE2 (1982): 23-250.

[3] U.S. EPA Municipal Environmental Research Laboratory, *Comparison of Leachate Characteristics from Selected Municipal Solid Waste Test Cells*, prepared by R.J. Wigh (1979).

[4] Ibid.

The velocity of ground water and the retardation caused by absorption of contaminants to earth materials govern the rate of constituent transport through the unsaturated and saturated zones. Retardation depends on contaminant characteristics, such as hydrophobicity and polarity, and on soil characteristics, including fraction of organic carbon, porosity, and bulk mass density. We considered degradation of chemicals where data are adequate to develop rates, and this factor can have a major effect on concentration profiles.

Exposure

The ground-water concentrations of chemical constituents released from landfills can cause human exposure through drinking water. We assumed that all exposed individuals weigh 70 kilograms and drink two liters of water per day. We calculated the lifetime dose as the running 70-year average over an individual's lifetime.

Impacts: Human Health Risk and Resource Damage

To estimate risks for carcinogenic effects we used the dose response functions developed by EPA's Carcinogen Assessment Group (CAG) (i.e., the upper 95 percent confidence interval on carcinogenic potency). For systemic effects, we used the Weibull equation with a threshold to predict a probability of effect, rather than the traditional direct comparison to some reference dose.

We report risks as the average, most exposed individual (MEI) risk (i.e., the average lifetime risk to the most exposed individual over the modeling period). By using data on the size of the population drinking from wells, we also predicted population risks.

Resource damage (RD) is an alternate measure of impacts associated with releases from Subtitle D landfills. It is based on the premise that ground water is a valuable resource, and that there is a loss in resource value if the water is rendered unfit for human consumption. We assigned a monetary value to the drinking water supply by assuming that it is no more valuable than the least expensive substitute. Thus, we estimated resource damage as the cost to provide alternative drinking water supplies to people whose wells are contaminated by releases from landfills.

Human health risk and resource damage are mutually exclusive measures of impacts. The former quantifies the potential risks from the ingestion of contaminated ground water, while the latter estimates the costs of replacing the contaminated water supplies. These impacts obviously could not occur simultaneously, because exposure would end when the water supply is replaced, and therefore one cannot add risk and resource damage to derive an overall estimate of impacts. Resource damage does provide us with an alternate measure of impacts, one that (unlike human health risk) provides some measure of benefits even if no people currently live near the landfill. Resource damage is a particularly important measure of benefits in the current analysis, where over half of the units pose no risk because there are no nearby wells.

We used the costs of replacing water supplies only as a way to place a monetary value on ground water; these replacement costs should not be viewed as corrective action costs. We assumed that ground water is unsuitable for human consumption when pollutant levels exceed regulatory limits (maximum contaminant levels, or MCLs) or taste and odor thresholds, and these levels do not necessarily correspond to those that trigger corrective action.

In our approach, RD depends on the area of the plume, the time it first appears, the drinking-water well density, the source of the alternative water supply, and whether or not the ground water is currently used. The Subtitle D Risk Model calculates the maximum plume area based on the maximum plume width at each of the seven well distances. The maximum plume area multiplied by well density and number of users per well determines the total population affected and sets the size of the replacement water supply system. The well density used in the analysis (one well per 80 acres) and the average population served per well (128) are the mean values reported in the Subtitle D facility survey for those facilities with private or public wells. To estimate water supply system size, we assumed that each person consumes 65 gallons of water per day (total use).

The replacement scenarios offer a choice between replacing contaminated ground water with nearby ground water, nearby surface water, distant ground water, or distant surface water. The two distances chosen to define the scenarios are one mile and 10.5 miles. Each scenario includes the costs to acquire, transport, treat (surface water only), and distribute water to the user population. For this analysis we used the "nearby ground-water" replacement scenario, which we believe is the most typical.

The last parameter determining resource damage is the use scenario. We estimated RD under two scenarios: use value and option value. Use value represents the RD for the population that is currently using the ground water for drinking water, whereas option value represents the RD for the population that is not currently using the ground water for drinking water but may wish to do so in the future. For use value, the RD equals the present value of the cost to replace the ground water. For option value, the RD measure incorporates the probabilistic nature of future use; replacement costs are multiplied by an estimated probability of use in each time period. These products are summed in present value terms.

Corrective Action

The Subtitle D Risk Model also simulates the effects of various corrective measures on contaminant concentrations and, consequently, on risk and resource damage. In the model, corrective actions are triggered when contaminant concentrations exceed specific thresholds (analytical detection limits or, for those pollutants regulated under the Safe Drinking Water Act, MCLs) at a downgradient monitoring well. The model employs decision criteria to select an appropriate corrective action technology (CAT) (e.g., capping, excavation, slurry walls, extraction wells, or drains) on the basis of landfill size, hydrogeology, and the distance the plume has traveled. The model estimates the effects of the selected corrective action on downgradient well concentration profiles and recomputes risk and resource damage. For this analysis we limited the model to the use of recovery wells as the appropriate CAT, since they were generally found to be the least expensive method.

Model Inputs and Supporting Analyses

The model runs used to analyze the effectiveness of the regulatory options cover a variety of design and environmental scenarios. We explain below our choice of model inputs, some of the supporting work on which the analysis is based, and our methods for assigning frequency weights to each set of factors.

Facility Size

We modeled three landfill sizes: 10 TPD, 175 TPD, and 750 TPD. Each category is characterized by the total volume of waste placed in the landfill, the number of phases used to dispose the waste (the common practice is to sequentially fill and then close individual sections of the landfill), and the dimensions of the landfill at capacity (surface area, depth, height). The waste volumes and dimensions for each capacity category were estimated by the design and cost model described in Chapter IV. The number of phases in the risk analysis are 2, 5, and 10 for the 10-TPD, 175-TPD, and 750-TPD landfills, respectively.

We used EPA's MSWLF survey to estimate the frequency with which each landfill size category occurs nationwide. We included landfills with capacities of up to 30 TPD in our 10-TPD category, 30 to 500 TPD in the 175-TPD category, and larger capacities in the 750-TPD category. Using this approach, we modeled 61.5 percent of the landfills as 10 TPD, 33.1 percent as 175 TPD, and 5.5 percent as 750 TPD. We assumed that facility size is independent of hydrogeologic and exposure attributes.

Facility Design

For the risk and resource damage analysis, we assumed that all landfills are new. For the baseline, we modeled a landfill that operates for 20 years and then closes with a vegetative cover. There is no post-closure care period, no ground-water monitoring, and no corrective action. To simulate the impact of the regulatory options, we assumed that a new landfill is constructed at the same site and is designed and operated in compliance with the regulatory option throughout its lifetime (20 years) and post-closure care period.

We did not model the effectiveness of the regulatory alternatives at existing units, where some of the waste is already in place; this would have required us to model pollutant release and transport at a unit that has mixed designs (e.g., some unlined cells and some lined cells), which is currently beyond the capability of the Subtitle D Risk Model. This also limited our ability to model corrective action costs for existing MSWLFs.

Unlike the cost analysis, the risk analysis did not take account of states with existing cover and containment system requirements that would satisfy the uniform or performance standards under the final rule. In 22 states, the incremental risk reduction or RD reduction attributable to these systems should not be credited to this rulemaking. To the extent that we have not accounted for existing state containment system requirements, we overestimate risk reduction somewhat under the final rule.

Our approach for simulating post-closure care and corrective action is described in the next two sections. In the rest of this section, we discuss how we analyzed facility design under each regulatory option.

Final Rule

The designs imposed on landfills under the final rule can range from the uniform design to an alternative design that complies with the performance standard. As in the cost analysis, we assigned the uniform design (composite liner, leachate collection system, synthetic cover) to about 27 percent of the landfills. These landfills were selected from scenarios characterized by wet climates, shallow ground-water settings, and rapid ground-water flow rates.

We assumed that the other 73 percent of MSWLFs would be subject to the performance standard. To analyze how these landfills would be designed, we ran the Subtitle D Risk Model using three different designs: unlined with vegetative cover, unlined with synthetic cover, and synthetic-lined with synthetic cover plus an LCS; each of these designs is described in Chapter IV. We used the maximum pollutant concentrations simulated at the 10-meter well (the Subtitle D Risk Model well distance used to represent the unit boundary) as the basis for assigning designs.[5] We assigned each size/infiltration/flow field scenario to the least stringent design capable of maintaining all pollutant concentrations below their MCLs at the 10-meter well. We assumed that all scenarios in which a constituent exceeded its MCL at the 10-meter well with the unlined/vegetative cover design would, at a minimum, install a synthetic cover at closure. If all pollutant concentrations at the 10-meter well were reduced to below their MCLs by installing the cover, we assumed that this design would be sufficient to meet the performance standard. For landfills that could not meet the performance standard with a synthetic cover, we assumed that both a synthetic liner (with an LCS) and a synthetic cover would be needed.

Alternative 1

Under the proposal, landfills are required to meet a design goal ranging between 10^{-4} and 10^{-7}. In our modeling, we assumed that the design goal would be set at 10^{-5} at the compliance point. As modeled this was interpreted to mean that constituent concentrations would not exceed levels that would impose a 300-year average individual lifetime cancer risk of 10^{-5} at the 10-meter well (used to represent the unit boundary).[6]

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[5] As will be discussed later, eight constituents of concern (COC) were simulated using the Subtitle D Risk Model. There are several other constituents with MCLs that were not simulated by the model. For many of them, we knew their leachate concentrations, and estimated their concentrations at the 10-meter well. Based on a comparison of physical and chemical properties (mainly retardation and degradation rates), we matched each constituent to one of the eight COCs that was actually modeled. We assumed that the matched pair would behave similarly in ground water, and that the ratio of their leachate concentrations to their maximum concentrations at the 10-meter well would be the same. Based on the known ratio of the leachate-to-ground-water concentration of the modeled constituent, and the known leachate concentration of the constituent that was not modeled, we estimated the maximum ground-water concentration of the latter and compared it to the MCL.

[6] Using the 300-year average individual lifetime risk may underestimate the number of scenarios in which the 10^{-5} threshold is exceeded and, therefore, underestimate both the fraction of units assigned the more expensive designs and the risk reduction provided by the proposal. At some time during the averaging period, constituent concentrations may exceed levels leading to a risk of 10^{-5}; to the extent that the design goal is set at a maximum lifetime or an "instantaneous" threshold, use of the average will smooth out these peaks.

To analyze the proposal, we simulated risk using three different designs: unlined with vegetative cover, unlined with synthetic cover, and synthetic-lined with synthetic cover plus an LCS.

As with the final rule, we assigned each size/infiltration/flow field scenario to the least stringent design capable of meeting the design goal. We assumed that all scenarios with 300-year average individual risk exceeding 10^{-5} at the compliance point would install a synthetic cover at closure. If 300-year average individual risk at the compliance point was reduced to 10^{-5} or less by installing the cover, we assumed that this design would suffice. For landfills that could not meet the performance standard with a synthetic cover, we assumed that both a synthetic liner (with an LCS) and a synthetic cover would be necessary.

Alternative 2

Under Alternative 2, all landfills are required to have a composite/synthetic liner, two leachate collection systems, and a composite cover. To analyze the effectiveness of Alternative 2, we simulated risk and resource damage for each size/setting/flow field/well distance scenario under this design. If constituent concentrations at the unit boundary were above MCLs, health-based limits, or background, then corrective action would be triggered.

Alternative 3

Alternative 3 imposes both design and performance standards on landfills. The only explicit design requirement is for leachate collection systems at landfills that generate more than one foot of leachate during their active lives. Performance standards relating to time of travel to the uppermost aquifer and to infiltration of water through the cover may be met with a number of different designs. As discussed in Chapter IV, we used some simplifying assumptions to identify seven distinct designs that are likely to emerge in response to Alternative 3.

Given resource constraints, seven designs were too many to simulate for the benefits analysis. We thus reduced the number of designs to four: unlined with a vegetative cover, clay-lined with a vegetative cover, unlined with a synthetic cover, and synthetic-lined with a synthetic cover. We used the synthetic/synthetic design as a proxy for two of the designs that we did not model rigorously (unlined, LCS, synthetic cover; and synthetic-lined, LCS, vegetative cover). Each of these designs is applicable to about 8 percent of all landfills. Both require an LCS combined with either a synthetic liner or a synthetic cover, and of the four designs that we modeled, the synthetic/synthetic (with LCS) design is most similar in terms of leachate release. We used the baseline design (i.e., unlined, no LCS, vegetative cover) as a proxy for the design that requires no liner, an LCS, and a vegetative cover. This design occurs infrequently (about 3 percent of the landfills).

Alternative 4

Under Alternative 4, landfills are allowed to operate with vegetative covers and no liners, but must perform corrective action if constituents exceed corrective action triggers at the unit boundary during the operating life or post-closure care period. For landfills that did not trigger corrective action in the baseline, we assumed that risk and resource damage was equal to baseline levels. Our approach for analyzing the triggering and effectiveness of corrective action is discussed below.

Post-Closure Care

Post-closure care activities that were analyzed in the risk model include maintenance of the cover (and replacement of the cover, if necessary), leachate collection (for designs with leachate collection systems), and ground-water monitoring. At the end of the post-closure care period, these activities cease simultaneously. In the original model runs performed for the risk and resource damage analysis, the duration of the post-closure care period was set at 30 years. Because of resource constraints, however, we were unable to perform new risk model runs to analyze rigorously the rule's flexibility regarding the duration of post-closure care. Nonetheless, our approach for adjusting the original model outputs to account for different post-closure care periods is discussed below.

Differences in the duration of post-closure care can affect risk and resource damage in several ways:

- The length of the post-closure care period sets the duration of ground-water monitoring. Under a 10-year post-closure care period, corrective action can be triggered only during the first 30 years (20-year active life plus 10-year post-closure care period), while releases from the landfill can trigger corrective action up to the 60th year if the post-closure care period is extended to 40 years. Determining how many facilities would trigger corrective action under different monitoring periods was accomplished simply by examining the corrective action trigger year, an output of the original model runs. As will be discussed in the next section, the proportion of landfills that trigger corrective action does not differ significantly for post-closure care periods ranging from 10 to 40 years.

- For designs with liners and covers, longer post-closure care means longer maintenance of containment systems, resulting in increased pollutant degradation prior to release and a delay in the onset of releases. Without new model runs, we were unable to develop an approach for quantifying the effects of increases (or decreases) in pollutant degradation on simulated ground-water concentrations. However, for resource damage, the delay itself (regardless of

> its effects on pollutant concentrations) is an important phenomenon. Unlike human health risks, we discount resource damage, expressing water supply replacement costs in present-value terms. For resource damage, then, a simple way to quantify the impact of changes in the duration of post-closure care was to adjust the discounting of the water supply replacement costs. We adjusted the original analysis of 30-year post-closure care by increasing the delay in the onset of release by 10 years (extending the post-closure care period from 30 to 40 years). Conversely, we reduced the time to releases by 20 years to model 10 years of post-closure care instead of 30. Delaying the onset of releases by extending the post-closure care period by 30 years (from 10 years to 40 years of care) would reduce resource damage by nearly 60 percent. This is the adjustment we made for landfills designed with liners and covers; it was not appropriate for landfills without liners, for which releases begin during the active life.

We did not adjust the risk results provided by the model to quantify how simulated risk would increase or decrease under shorter or longer post-closure care periods. Instead, we report risk for 30 years of post-closure care and address changes in the duration of the post-closure care period qualitatively.

Corrective Action

For the purposes of this analysis, we assumed two different trigger levels for corrective action, depending on the regulatory option.

- For the final rule and all alternatives except Alternative 1, corrective action is triggered at (1) the MCL, for constituents with MCLs; or (2) for constituents without MCLs, the higher of either a health-based threshold (10^{-5} risk level for carcinogens, reference dose for noncarcinogens) or an analytical detection limit.
- For Alternative 1, the 1988 proposal, corrective action is triggered at the 10^{-5} risk level.

Since the corrective action triggers in the original Subtitle D Risk Model runs were based on MCLs or detection limits, we describe below how we adjusted those model runs to approximate the costs and effectiveness of corrective action under the final rule and regulatory options.

New Units

To determine which scenarios would trigger corrective action under the final rule and all alternatives except Alternative 1, we compared the maximum concentrations of all eight constituents of concern to the new corrective action triggers. As in the original model runs, we used the 10-meter well as the compliance point. This comparison was made off-line, using the maximum pollutant concentrations simulated by the model. For Alternative 1, we assumed that corrective action would be triggered if 300-year average individual risk at the 10-meter well exceeded the design goal of 10^{-5}. To determine the corrective action costs, risk, and resource damage for each scenario in which corrective action is triggered, we used the original model results.

Some interesting points regarding corrective action under the final rule and regulatory alternatives are:

- The post-closure care standards in the final rule require cover maintenance, leachate collection, and ground-water monitoring; we analyzed this same approach to post-closure care for all of the regulatory options. In our modeling, we make the simplifying assumption that for landfills designed with liners and covers, leachate collection and cover maintenance continue until the end of the post-closure care period. However, ground-water monitoring also stops at the end of post-closure care, so releases that occur after leachate collection and cover maintenance cease do not trigger corrective action at the monitoring wells. Therefore, corrective action is never triggered, in our modeling, for landfills designed with synthetic liners and synthetic covers. As a result, corrective action is never triggered for new units except under Alternative 4.[7]

- Corrective action is never triggered under the final rule or Alternative 1 because the design goal and the corrective action triggers are the same. Scenarios assigned to the unlined/vegetative cover or unlined/synthetic cover design do not trigger corrective action because if pollutant levels in ground water had exceeded MCLs or a 10^{-5} risk level, the scenarios would have been assigned a more stringent design; and, as discussed above, scenarios with synthetic liners and synthetic covers do not trigger corrective action in our modeling under any of the regulatory options.

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[7] This differs significantly from the 1988 draft RIA for the proposed rule, in which we simulated a two-phase post-closure care period for all regulatory alternatives. Because ground-water monitoring continued after cover maintenance and leachate collection ceased, landfills designed with liners and covers frequently triggered corrective action in the 1988 draft RIA.

- For landfills designed with only vegetative or synthetic covers, the portion that triggers corrective action does not change substantially when the post-closure care period is increased from 10 to 40 years (i.e., when the ground-water monitoring period ends at the 30th versus the 60th year). In other words, in our modeling, most landfills that trigger corrective action do so before the 30th year. This is because (1) we made the conservative assumption that releases from the landfill begin immediately, rather than after the moisture storage capacity of the waste is reached; and (2) the compliance point is assumed to be at the 10-meter well, which is very close to the landfill.

Existing Units

Corrective action at existing units was modeled differently than for new units, for two reasons. First, we assumed that states will decide to react to existing contamination more passively than they will for future contamination. This is primarily because of practical constraints (money, technical resources) on addressing existing releases. Second, modeling the performance of existing units with starting dates ranging from 1986 back to the late 1800s is inherently difficult.

Existing units that trigger corrective action were modeled as having two possible responses. For those with the most extensive contamination (i.e., the contaminants imposed risk greater than 10^{-5} at the 600-meter well distance), active cleanup was required. The remaining landfills were not assigned cleanup costs, but were instead assigned costs to replace the contaminated ground-water supply, assuming flexibility in the timing of the cleanup. For these remaining landfills, if resource damage is triggered in the Risk Model, the owner and/or operator must replace the ground-water supply at that time. If wells exist downgradient, we applied replacement costs using the same algorithms applied in the use value approach to measuring resource damage (i.e., the wells are currently used as a drinking water source). Those without wells pay replacement costs in the future, based on the same approach used to estimate option value (i.e., the costs are computed on an expected value basis to account for future use of the water).

The difficulty with modeling corrective action at existing facilities is the range of ages of landfills. To account for the different ages, we categorized landfills into age categories and calculated the incidence and costs for corrective action given several discrete estimates for past life.

Because all existing units are assumed to have similar designs for the benefits analysis, we assumed that existing corrective action costs would be the same across all the regulatory options. The addition of more elaborate cover systems at closure would probably reduce corrective action costs compared to those presented here, but time and model constraints prevented our analyzing this in any detail. This will tend to overestimate corrective action costs somewhat.

Environmental Settings

The timing of pollutant release and the rate at which pollutants enter an aquifer are influenced significantly by climate and hydrogeology. Two important parameters are net infiltration (precipitation minus evapotranspiration) and ground-water table depth. Net infiltration determines the amount of water that can enter a landfill as a result of precipitation. Ground-water table depth is important for two reasons. First, the depth to ground water determines the thickness of the unsaturated zone, an area in which significant pollutant retardation and degradation may occur. Second, for facilities that are seasonally inundated with ground water, the inundation depth determines the rate at which ground water can flow through the waste.

We used these two parameters to define eight environmental settings for the failure/release submodel runs. The settings, shown in Table VI-1, consist of four annual net infiltration regimes (0.25-inch, l-inch, 10-inch, and 20-inch) and two categories of ground-water table depths for each infiltration region (deep and shallow). To determine the mean depth to ground water and the average annual ground-water fluctuations, we performed a statistical analysis of U.S. Geological Survey (USGS) data for each infiltration category. We chose approximate 50th and 90th percentile water table depths to represent the shallow and deep conditions, respectively.

Table VI-1

Failure/Release Submodel
Environmental Settings

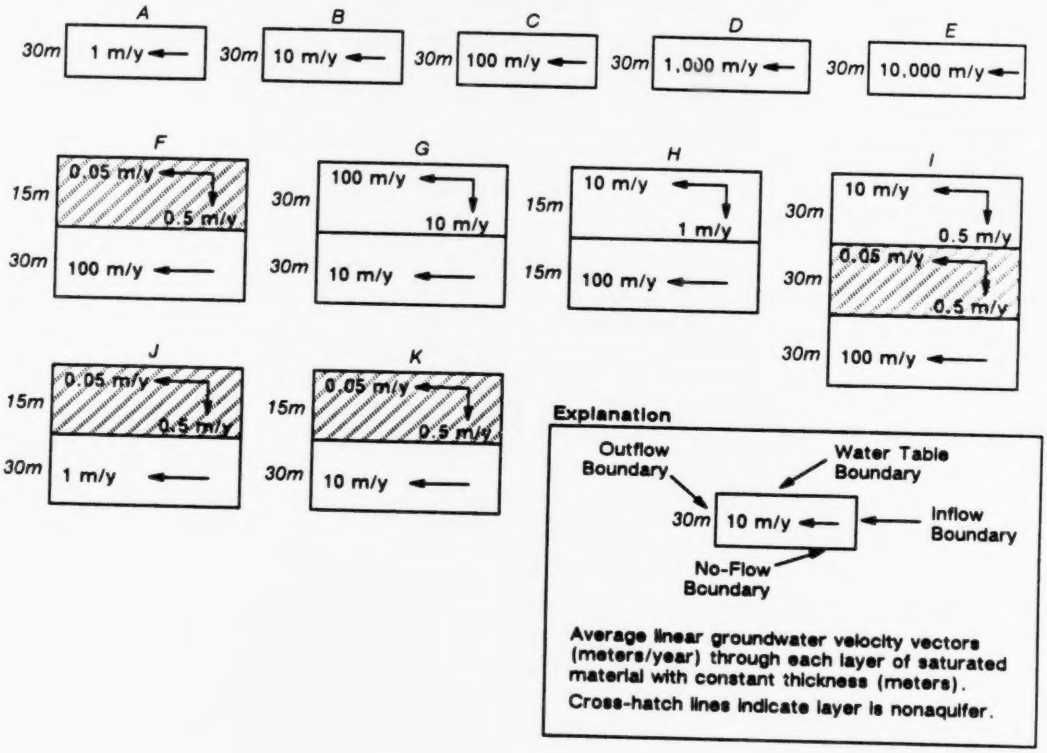
Annual Net Infiltration (Inches)	Mean Water Table Depth/ Average Annual Fluctuation (feet)
0.25	Deep: 200/1.7
0.25	Shallow: 17/1.7
1.0	Deep: 129/1.8
1.0	Shallow: 20/1.8
10.0	Deep: 47/2.4
10.0	Shallow: 15/2.4
20.0	Deep: 52/2.8
20.0	Shallow: 20/2.8

For all eight settings, we assumed that the subgrade underlying the waste has a saturated hydraulic conductivity of 10^{-4} centimeters per second (cm/sec). This subgrade is sufficiently permeable to allow leachate to be released at the same rate at which it is formed. For most net infiltration regions, the subgrade conductivity would have to be very low (i.e., 10^{-7} cm/sec or less) before it would limit leachate release rates. According to EPA's MSWLF survey, few landfills have subgrade conductivities this low.

To model the transport of constituents in the saturated zone, we developed 11 generic ground-water flow fields to represent the range of hydrogeologic conditions in the United States. We based the flow fields on data collected from ground-water supply reports for each of the USGS ground-water regions. The flow fields, shown in Figure VI-1, vary in terms of aquifer configuration, materials, and flow velocity. Five of the flow fields (A, B, C, D, and E) are single-layer aquifer systems, two (G and H) contain two adjacent aquifers, three (F, J, and K) consist of an aquifer overlain by a nonaquifer, and one (I) contains two aquifers separated by a nonaquifer.

Figure VI-1

Eleven Generic Ground-Water Flow Fields
Used in Saturated Zone Transport Modeling



Average linear groundwater velocity vectors (meters/year) through each layer of saturated material with constant thickness (meters).
Cross-hatch lines indicate layer is nonaquifer.

To estimate the frequency with which landfills are located in each of these environmental settings, we determined the latitudes and longitudes of over 700 landfills from the facility survey and a separate mapping effort. We used precipitation and other climatic data collected at weather stations near each landfill to assign the site to a net infiltration region. We also determined the likely DRASTIC setting[8] for each landfill, and used ground-water data characterizing these settings to assign sites to ground-water table depths and flow fields. Flow fields I and J were virtually absent from our set of characterized facilities, and therefore we did not model them.

We assigned each environmental setting (i.e., combination of net infiltration, depth to water table, and flow field) a unique frequency weight. These weights were used to scale the risk-model weighted corrective action costs described earlier in this chapter. We assumed that the environmental setting was independent of facility size and exposure characteristics.

Leachate Characterization

We simulated one representative leachate characterized by a specified set of constituents. This typical leachate is intended to represent leachates generated from the co-disposal of municipal solid waste, nonhazardous industrial waste, household hazardous waste, and small quantity generator hazardous waste. Because leachates for similar landfills vary significantly in composition, we determined characteristic constituents (and their initial concentrations) by analyzing leachate composition data covering 212 chemical constituents at 44 operating MSWLFs.

The landfills included in the leachate characterization analysis were the OSW case study landfills, New Jersey landfills that discharge leachates under National Pollutant Discharge Elimination System (NPDES) permits, landfills surveyed for the presence of organic chemicals in Minnesota, and landfills submitting leachate sampling results to the Wisconsin Natural Resources Department. We determined a representative concentration of each constituent by calculating its median value at each landfill (if sampled more than once at a landfill), and then its median value across all landfills. We used median rather than mean concentrations in order to reduce the effect of anomalous values. In determining median concentrations, we used the detection limit as default concentrations for situations in which a constituent was tested for but not detected. We calculated median concentrations for 96 of the 212 constituents that had been analyzed at the 44 facilities; we did not calculate median concentrations for 116 constituents because they either had not been detected at any facility or had been tested for at only one facility.

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[8] U.S. EPA, *DRASTIC: A Standardized System for Evaluating Ground Water Pollution Potential Using Hydrogeologic Settings*, EPA/600/2-85/018 (1985).

From the final list of 96 constituents observed in MSW leachate, we selected eight constituents of concern (COC) for risk modeling. Our approach for selecting a COC was designed to identify the constituents with the highest potential for causing human health risk or resource damage on the basis of the following considerations:

- Observed concentrations in MSWLF leachate
- Toxicity to humans
- Regulatory limits under the Safe Drinking Water Act
- Taste and odor thresholds
- Mobility and persistence in the subsurface environment

To select the COC, we began with the medians of the observed constituent concentrations in leachate. Borrowing from the modeling efforts of the Organic Toxicity Characteristic Rule (FR 51:21648), we divided these by a conservative dilution/attenuation factor of 10 (only 10 percent of landfills would have less dilution) to estimate concentrations in ground water at a well 500 feet from the landfill. We compared these ground-water concentrations with risk-based threshold values for each constituent. The human health risk threshold corresponds to a cancer risk of 10^{-6} for carcinogens or a reference dose for noncarcinogens. The resource damage threshold is the lowest of a Safe Drinking Water Act MCL (existing or proposed), a taste and odor threshold (from the Liner Location Model database), or a threshold for agricultural use (EPA's Water Quality Criteria).

To make the comparison, we divided the ground-water concentration by both the human health risk and resource damage threshold for each constituent. We eliminated from further consideration all ratios below 1, leaving 22 constituents as candidates for risk modeling. We selected the final eight COCs from this list, giving primary consideration to the magnitude of the ratio, but also considering secondary factors such as mobility and persistence.

Table VI-2 presents the eight COCs, the effect for each, and the ratio of its median concentration in ground water to its threshold concentration.

Table VI-2

Constituents of Concern

Constituent	Ratio[1]	Criterion (effect)
Vinyl chloride[2]	658	Human health risk (cancer)
Arsenic	579	Human health risk (cancer)
Iron	22	Resource damage (taste and odor)
1,1,2,2-Tetrachloroethane	11	Human health risk (cancer)
Dichloromethane	9	Human health risk (cancer)
Carbon tetrachloride	4	Human health risk (cancer)
Antimony	3	Human health risk (systemic)
Phenol	2	Resource damage (taste and odor)

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[1] Median concentration in ground water divided by the threshold concentration.

[2] Because tetrachloroethene, trichloroethene, and 1,2-dichloroethene degrade into vinyl chloride, we took the stoichiometric equivalents of the three constituents and converted them to concentrations of vinyl chloride. Because vinyl chloride is more toxic than the other three chloroethenes, this approach is conservative.

After we completed the risk and resource damage modeling, additional leachate data were made available for 55 landfills that were not included in the original database. We derived new medians as well as 90th percentile leachate concentrations based on the revised database, which now comprises 99 landfills. The results are presented in Table VI-3. As shown, the revised median values are very close to the median concentrations used in the model (the greatest difference is for antimony, whose revised median value is one-eighth the original). The median value of vinyl chloride is 50 percent higher based on the revised database, and the median value of iron is slightly higher. For all other constituents, the revised median values are equal to or less than the values simulated in the model. We did not simulate risk or resource damage using these revised median levels. However, based on a comparision of the median concentrations, we believe that the results would not have changed significantly had the revised medians been simulated in the model. The revised 90th percentile concentrations are also quite close to those calculated based on the 44 original landfills. The largest difference is for phenol, whose revised 90th percentile concentration is about one sixth the original 90th percentile. This confirms that the leachate data used for the RIA risk and resource damage analysis are representative of the more complete set of data available today. In Chapter IX, we place an approximate upper bound on risk using the revised 90th percentile concentrations listed in Table VI-3.

Table VI-3

Median and 90th Percentile Concentrations for Constituents of Concern

(milligrams/liter)

Constituent	Original Database Median	Original Database 90th Percentile	Revised Database Median	Revised Database 90th Percentile
Vinyl chloride[1]	0.10	0.28	0.15	0.47
Arsenic	0.01	0.02	0.01	0.03
Iron	66.2	519	67.3	516
1,1,2,2-Tetrachloroethane	0.02	0.13	0.007	0.04
Dichloromethane	0.23	2.21	0.15	1.51
Carbon tetrachloride	0.01	0.013	0.005	0.02
Antimony	0.47	0.88	0.06	0.65
Phenol	0.26	8.73	0.04	1.52

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[1] Because tetrachloroethene, trichloroethene, and 1,2-dichloroethene degrade into vinyl chloride, we took the stoichiometric equivalents of the three constituents and converted them to concentrations of vinyl chloride. Because vinyl chloride is more toxic than the other three chloroethenes, this approach is conservative.

Impacts

Of the constituents of concern selected for modeling human health risk, five are carcinogens and one is a noncarcinogen. Our approach for estimating risks for carcinogenic effects is consistent with EPA's cancer risk assessment guidelines and uses a function that is linear at low doses, with no threshold. We used the carcinogenic potencies developed by EPA's Carcinogen Assessment Group (i.e., 95 percent upper-bound slopes based on the linearized multistage model). The potencies used in the model are shown in Table VI-4.

Table VI-4

Carcinogenic Potencies Modeled for Constituents of Concern

Constituent	Potency $(mg/kg\text{-}day)^{-1}$
Arsenic[1]	15
Carbon tetrachloride	0.13
Dichloromethane	0.0075
1,1,2,2-Tetrachloroethane	0.2
Vinyl chloride	2.3

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[1] The carcinogenic potency for arsenic has recently been lowered to $2(mg/kg\text{-}day)^{-1}$. Because arsenic rarely poses significant risk in our model runs, and never causes resource damage, this change has minimal impacts on our results.

For noncarcinogenic effects, we used the Weibull equation with a threshold to predict a probability of effect. Below the threshold, risk equals zero. At doses above the threshold, risk depends on the dose, the constituent-specific threshold, and the dose-response parameter. Antimony is the only systemic toxicant selected for risk modeling; the threshold and dose-response parameter used for antimony are 4.1×10^{-5} (milligrams per kilogram-day [mg/kg-day]) and 1.4×10^{5} $(mg/kg\text{-}day)^{-2}$, respectively. Predicted antimony concentrations never exceed the threshold in our model runs, so all the risk predicted is cancer risk.

Exposure Distance and Populations

We selected seven well distances for modeling risk: 10 meters, 60 meters, 200 meters, 400 meters, 600 meters, 1,000 meters, and 1,500 meters. We used data from EPA's MSWLF survey to develop a distribution of distance from the landfill to the closest well at each site, and used this well distance distribution to develop frequency weights for each of the seven wells (Table VI-5). Because risk usually decreases with increasing distance from the landfill, we used this distribution (distance to the *closest* well) to estimate risk to the most exposed individual (MEI). Note that only 46 percent of the landfills reported that they currently have downgradient wells within one mile. Thus, at the other 54 percent of the facilities, there would be no human health risk from drinking ground water within one mile of the unit.

Table VI-5

Distance to Nearest Public or Private Well

(meters)

Model Well Distance	Actual Well Distance Range	Percentage of Landfills
10	0-35	2.4%
60	35-130	4.4
200	130-300	6.0
400	300-500	12.7
600	500-800	5.0
1,000	800-1,250	9.8
1,500	1,250-1,600	5.3
None	Within 1 mile	54.4
		100.0%

Source: EPA Municipal Landfill Survey.

To calculate population risk (number of predicted cases), we used survey data on distance to *all* wells within one mile downgradient and the number of people served at each well. Across private and public wells, the mean well density for landfills with downgradient wells was about one well per 80 acres, with private wells occurring far more frequently. The mean population served per well was about 128, but this mean is strongly influenced by relatively infrequent but large populations serviced by public wells. These numbers indicate a density of about 1.6 well-using people per acre of downgradient land. We calculated the land area associated with each exposure well and multiplied the area by this population density to estimate the size of the exposed population for each well at sites with wells.

The results of the risk and resource damage modeling are described in Chapter IX.

VII. Results: Cost Analysis

In this chapter, we present the results of the cost analysis as well as the adjustments to costs, which take account of future shifts in waste management practices. We begin the cost analysis by examining the current expenditures of MSWLF owners and operators. These expenditures provide a benchmark against which to compare changes in operating costs due to regulation. In the analysis, we use the Design and Cost Model to estimate baseline costs so assumptions are consistent across the baseline and the five regulatory options. We then apply the cost methodology described in Chapter IV to estimate the incremental costs of compliance with the final rule and with the four regulatory alternatives. In the first section of the cost analysis, however, we compare this analytical baseline estimated using the Design and Cost Model with survey information on reported annual expenditures.

In subsequent sections of the cost analysis, we report costs as incremental costs above the analytical baseline. We examine the costs in three different ways: the costs for new landfills assuming a 20-year life as described in Chapter IV, the costs for existing units where the remaining life is variable, and the combined costs incorporating both new and existing unit costs. We also discuss corrective action costs that depend on risk model results. Because the risk analysis is limited to new units, we will use costs for new landfills to compare costs and benefits among the various options in Chapter X. In other words, the combined costs used to show national costs are not directly comparable to risk and resource damage results. For the economic impact analysis, however, we focus on the combined cost as a more complete measure of incremental costs attributable to the regulation.

The final part of the chapter reports the results of our analysis of regionalization, recycling, source reduction, and waste combustion as they affect future waste disposal and, therefore, compliance costs. The methodology for this analysis is presented in Chapter IV.

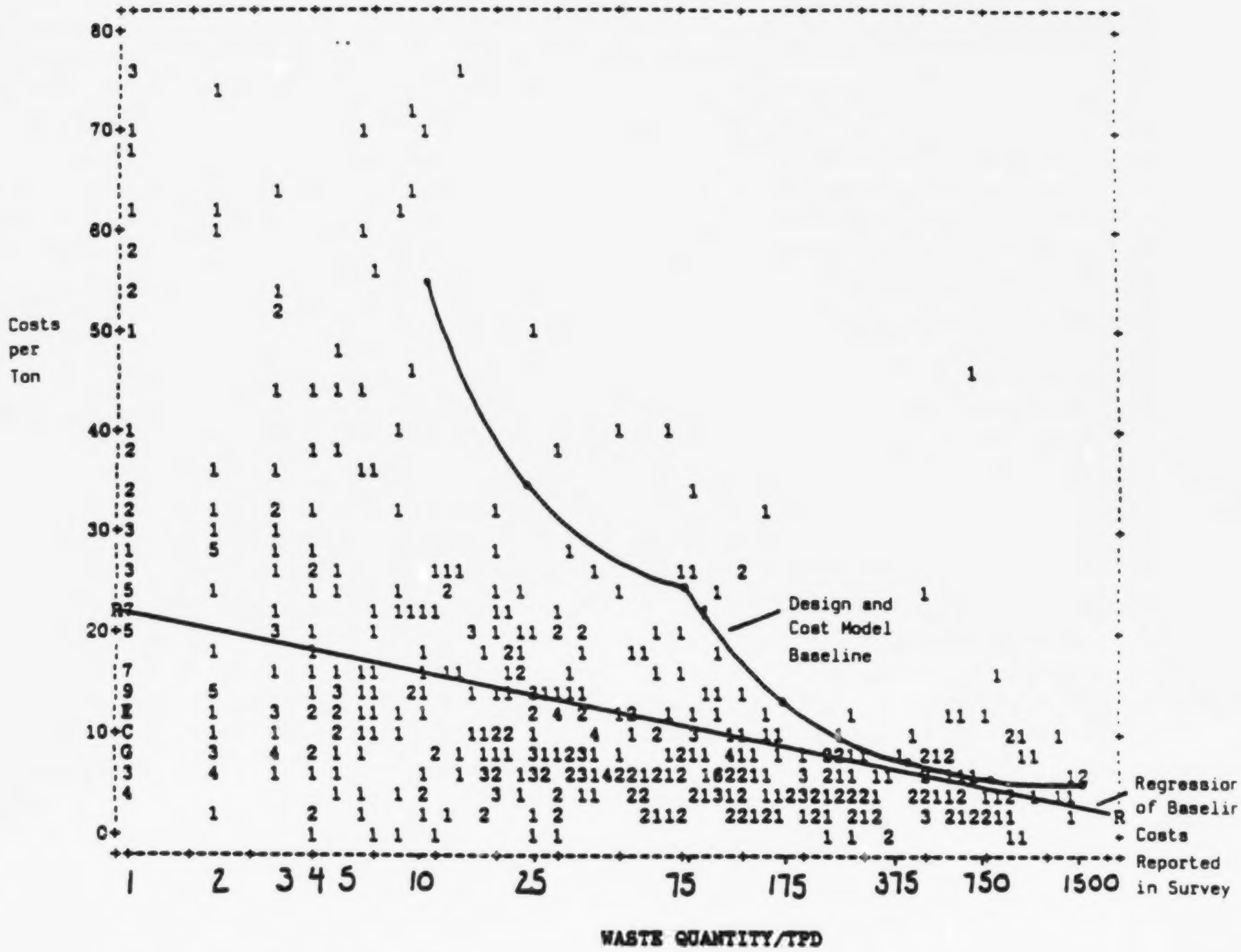
Baseline

The costs presented in this chapter are all reported as incremental costs: the difference in cost incurred because of the imposition of new regulatory requirements. To estimate the baseline from which to measure the difference, we used the Design and Cost Model described in Chapter IV. Throughout the RIA we refer to this as the analytical baseline.

As a validation exercise, we reviewed information from the facility survey reporting landfills' actual annual operating expenditures. Although these costs may not include amortization expenses for capital equipment, we adjusted the reported costs to costs per ton and plotted them against waste quantity received in tons per day. Figure VII-1 shows the resulting plot for the landfills that provided the estimates. Note that waste quantity is shown on a logarithmic scale. The numbers on the plot indicate the number of responses at each point from 1 through 9. "A" stands for 10, "B" 11, and so on.

Figure VII-1

Baseline Costs: Modeled Values Versus Reported Values



570 cases plotted. Regression statistics of CPT on LNWQ:
Correlation -.40928 R Squared

The figure emphasizes the clustering of landfills with small quantities accepted. The median cost per ton for all landfills surveyed is $11; this compares with a baseline median cost per ton of $35 estimated for all MSWLFs using the cost model.[1] As size increases, reported annual operating costs per ton decline. For reference, we fit a regression line to the reported data and included it on the figure.

In addition, we also plotted the cost model analytical baseline. While it lies above the regression line, especially for smaller units, it is within the range of reported estimates. Because the reported costs may not include equipment and other amortized capital expenses, we believe that the analytical baseline is a reasonable baseline for the analysis. In several subsequent sections of the chapter we will report incremental costs as a percentage increase above current annual operating costs; this refers to the increase above the reported costs, from the survey, plotted on this figure. Table VII-i shows the baseline cost estimates from the model for new landfills by size. These costs per ton do not include costs for state regulatory requirements. If we use the cost model to predict baseline national costs, the 6,034 landfills have an annualized cost of just over $2.1 billion (not including state requirements). As described in Chapter IV, the state regulations we analyzed add another $450 million, for a total baseline cost of $2.6 billion.

Table VII-i

Baseline Costs for New Landfill

(dollars per ton)

Size (TPD)	Baseline Cost
10	$54.95
25	34.46
75	23.93
175	12.83
375	8.52
750	6.25
1,500	6.23

Note: Costs do not include costs for state regulatory requirements.

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[1] This cost does not reflect costs incurred by landfill owners in states with ground-water monitoring or containment system requirements. These cost credits *are* taken into account in estimates of incremental compliance costs, however.

Final Rule

The final rule imposes a set of location and operating standards on all MSWLFs, regardless of their setting or design. These requirements are listed in Chapter IV. The compliance costs are more sensitive to the design assignments, which depend on compliance with the performance standard (if applicable) or on the uniform design standard. As described in Chapter IV, we implemented our own criteria to simulate the decisions of landfill owners as they respond to the requirements of the final rule. We reflected some of the flexibility provided for in the rule by incorporating into our estimates a range of post-closure care periods (from 10 to 40 years). In addition, we did not impose compliance costs for ground-water monitoring wells, containment systems, or cover systems for certain landfills as described in Chapter IV.

The results we present in this chapter first address the unit costs of compliance for landfills of different sizes assigned to different designs. These unit costs do not reflect the exemptions from requirements for some landfills; these exemptions *are* reflected in the calculation of combined costs and national costs.

Unit Costs for New Landfills

The costs for new landfills assume that construction begins immediately and that the units have a 20-year operating life. In computing combined costs and national costs, we delay the starting date until the remaining life (if any) of the existing landfill is exhausted. Table VII-1 presents the annualized incremental costs per ton for the design options we used to model compliance with the final rule. As described in Chapter IV, the uniform designs are assigned to landfills with a high potential for generating leachate that could escape quickly from the unit. The costs for this uniform design are higher than for any of the designs we modeled under the performance standard. The unit costs for a given size of landfill can vary by as much as an order of magnitude, depending on the design that is assigned.

These estimates demonstrate the positive returns to scale for MSWLF design and operation over most of the size range. At the largest units, the economies of scale break down for the designs with synthetic or composite liners. The incremental cost for these designs is dominated by the containment system, which is keyed to the area of the landfill. Since the depth of these model-sized landfills is fixed, the units get higher and wider as the tonnage increases. At larger sizes, however, the height approaches a practical limit. At that point, costs related to area increase rapidly with increases in tonnage. As a result, costs per ton may level out or even increase. In general, if tonnage increases but the area increases by a smaller percentage, the costs per ton of waste will decline. If area increases by the same or greater proportion, the unit costs will increase.

Table VII-1

Final Rule: Incremental Costs for New Landfills

40-Year Post-Closure Care Period
(dollars per ton)

Size (TPD)	Unlined/ Vegetative Cover	Unlined/ Synthetic Cover	Synthetic Liner/ Synthetic Cover	Composite Liner/ Synthetic Cover (on-site clay)	Composite Liner/ Synthetic Cover (off-site clay)
10	$16.10	$20.28	$37.22	$43.13	$50.12
25	9.48	12.91	26.60	31.58	37.31
75	4.71	7.03	16.04	19.43	23.30
175	3.20	5.05	12.16	14.87	17.93
375	2.04	3.46	8.87	10.94	13.28
750	1.21	2.21	6.01	7.47	9.12
1,500	0.91	2.02	6.25	7.89	9.73

***Percentage Increase Over Baseline*[1]**

Size (TPD)	Unlined/ Vegetative Cover	Unlined/ Synthetic Cover	Synthetic Liner/ Synthetic Cover	Composite Liner/ Synthetic Cover (on-site clay)	Composite Liner/ Synthetic Cover (off-site clay)
10	29.3%	36.9%	67.7%	78.5%	91.2%
25	27.5	37.5	77.2	91.6	108.3
75	19.7	29.4	67.0	81.2	97.4
175	24.9	39.4	94.8	115.9	139.8
375	23.9	40.6	104.1	128.4	155.9
750	19.4	35.4	96.2	119.5	145.9
1,500	14.6	32.4	100.3	126.6	156.2

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[1] Assumes no state regulations in baseline.

The percentage increases (compared to the baseline) range from less than 15 percent to more than 150 percent. These overstate actual increases, however, because the baseline does not include state requirements that already impose some of these costs.

The least expensive design is the unlined/vegetative. The dominant component of the cost under this design is for ground-water monitoring. This requirement includes semiannual monitoring during the active life as well as during post-closure, capital costs for wells, and hydrogeologic studies to determine the placement of wells. For a 10-TPD landfill that incurs all of these costs, they contribute 78 percent of the $16.10 incremental cost per ton shown in Table VII-1. The balance of the cost includes gas monitoring, run-on and run-off controls, and higher engineering and quality assurance fees on the excavation and design of the landfill. For 1,500-TPD landfills, the ground-water monitoring requirements account for 52 percent of the incremental compliance costs for this design; the balance is for general facility standards.

The costs for the unlined/synthetic design are still dominated by ground-water monitoring for the small landfills, but for the larger units, the cost of the cover is more significant. At a 10-TPD unit, the ground-water monitoring requirement accounts for 62 percent of the incremental costs, given a 40-year post-closure care period, while the cover accounts for another 21 percent. At a 1,500-TPD landfill, however, the cost of the cover is 55 percent of the incremental cost, with monitoring accounting for only 23 percent. As discussed above, the cover costs do not exhibit large returns to scale because of the depth limitation imposed on units in the analysis.

The cost of the containment system dominates the unit costs for all of the remaining designs on Table VII-1. For the 10-TPD units, the synthetic liner, leachate collection system, and leachate treatment costs account for 46 percent of the incremental costs. If a composite liner is required, the containment system accounts for 53 percent of the cost if clay is available on site, and 60 percent of the cost if the clay is brought from off site. Ground-water monitoring accounts for 34 percent of the cost for the synthetic/synthetic design, the cover adds another 11 percent, and the general facility standards account for the remaining 9 percent. For the most expensive design, composite/synthetic with off-site clay, the cover system adds 8 percent to the 60 percent resulting from the containment system; ground-water monitoring adds another 25 percent.

At the largest landfills, the containment systems are even more significant. For the synthetic/synthetic design, the containment and leachate collection systems contribute 68 percent of the cost with the cover adding another 18 percent. With a composite/synthetic design and off-site clay, the containment system is 79 percent of the total with the cover adding 11 percent more.

If states chose to allow shorter post-closure care periods for landfills, the costs of post-closure care and monitoring would decrease. Table VII-2 shows the cost difference between a 10-year period and a 40-year period. The difference in post-closure costs is quite small compared to the other compliance costs. For example, a 10-year post-closure care period instead of a 40-year period would reduce the costs for an unlined/vegetative design by 16 percent for a 10-TPD unit ($2.64 from Table VII-2 divided by $16.10 from Table VII-1). At a 1,500-TPD landfill, the savings would be only 10 percent. Since all of the other designs have higher compliance costs, this savings would decline as a percentage of total costs.

Table VII-2

Final Rule: Incremental Cost Between 10 and 40 Years of Post-Closure Care for New Landfills

(dollars per ton)

Size (TPD)	Difference In Cost per Ton
10	- $2.64
25	- 1.46
75	- 0.58
175	- 0.34
375	- 0.20
750	- 0.12
1,500	- 0.09

Later in this chapter we will report national costs for the final rule using both assumptions about post-closure care periods. First, however, we present incremental costs for existing MSWLFs and the combined cost estimates for both existing landfills and their replacement units.

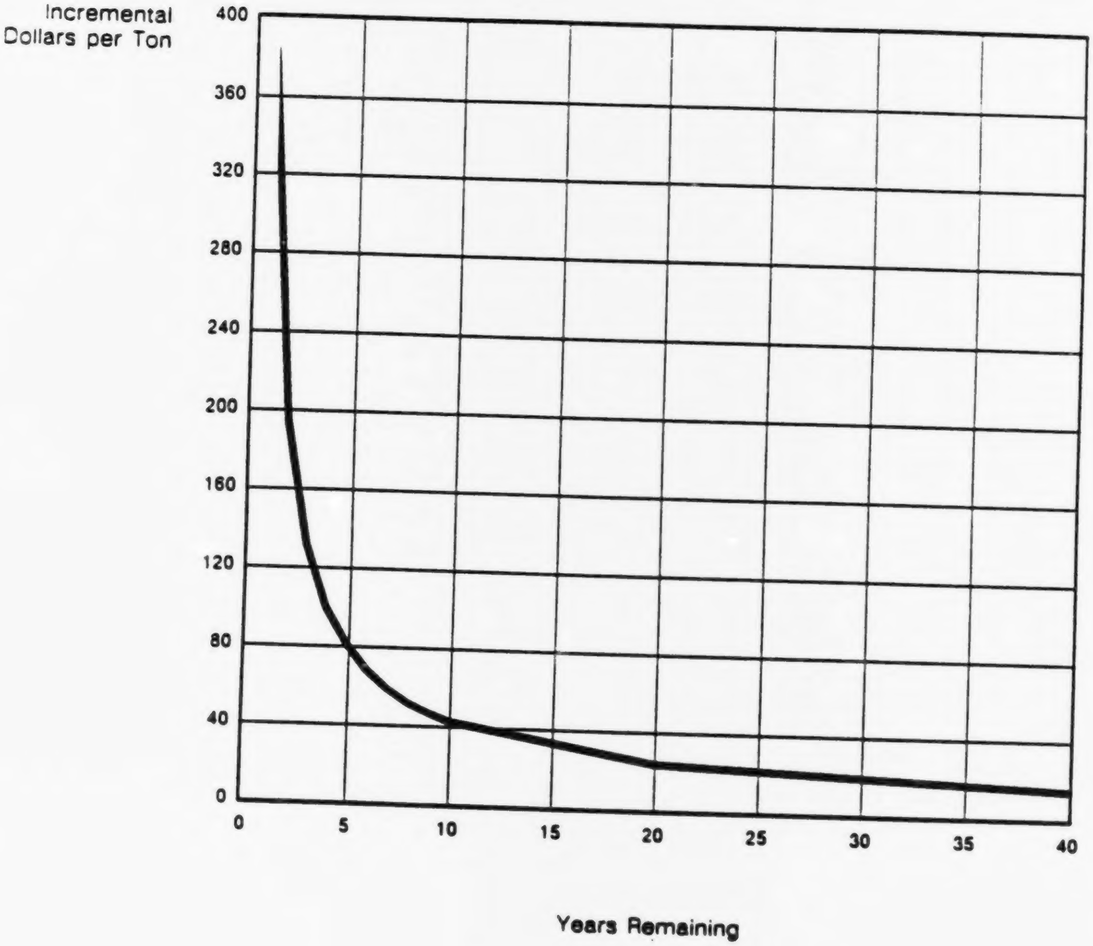
Unit Costs for Existing Landfills

The costs for existing units are very sensitive to the remaining life. In Figure VII-2 we show the effect of varying the remaining life given a $1 million expenditure at a 10-TPD landfill. The costs range from a high of $385 per ton with one year remaining to a theoretical minimum of $11.54 per ton. In the combined cost estimates presented later, we spread the costs for existing units over the actual remaining lives of the landfills. As an exercise to allow us to analyze the unit costs across the different design options for existing landfills, however, we chose to fix the remaining life at 10 years. Therefore, we can describe the incremental costs of the regulatory requirements without also showing the effect of varying remaining life. The impact analysis and the combined costs, however, will account for actual remaining life as reported in the survey.

Figure VII-2

Effect of Varying Remaining Life

$1 Million Cost at a 10-TPD Landfill



We provided only two different designs for existing units, one with a vegetative cover (where net precipitation was zero or negative) and one with a synthetic cover (where net precipitation was positive). Table VII-3 lists the unit costs for these designs for all sizes and for both post-closure care scenarios.

Table VII-3

Final Rule: Incremental Costs for Existing Landfills
10- and 40-Year Post-Closure Care Periods

(dollars per ton)

	10-Year Post-Closure		40-Year Post-Closure	
Size (TPD)	Unlined/ Vegetative Cover	Unlined/ Synthetic Cover	Unlined/ Vegetative Cover	Unlined/ Synthetic Cover
10	$20.57	$30.35	$26.75	$36.53
25	12.57	20.60	16.00	24.03
75	6.75	12.20	8.10	13.55
175	4.77	9.10	5.58	9.91
375	3.12	6.45	3.59	6.92
750	1.85	4.21	2.14	4.50
1,500	1.43	4.04	1.65	4.25

The difference in the post-closure care period is more significant for existing facilities because the monitoring that occurs during post-closure is a much greater share of total costs than it is for new facilities. The addition of 30 years to the post-closure care period increases costs by 5 to 30 percent, depending on size and design. If we assume a 40-year post-closure care period, the largest cost component for small landfills is always ground-water monitoring. At the 10-TPD size, monitoring accounts for 80 percent of the incremental cost for the unlined/vegetative design and 58 percent of the cost for the unlined/synthetic design. The cover accounts for 27 percent of the cost of the unlined/synthetic design. As with new units, the cover was more significant for the larger landfills. At 1,500-TPD landfills with a synthetic cover, the cover accounts for 61 percent of the incremental cost per ton. Ground-water monitoring is 48 percent of the unlined/vegetative cost and 19 percent of the unlined/synthetic cost.

Assuming 10 years of post-closure care, ground-water monitoring contributes 75 percent of the unlined/vegetative cost and 51 percent of the unlined/synthetic cost. The synthetic cover accounts for 32 percent of the unlined/synthetic design. For the 1,500-TPD units, monitoring is 44 percent of the unlined/vegetative design and only 16 percent for units with synthetic covers. The synthetic cover alone accounts for 64 percent of the unlined/synthetic design.

Combined Costs and National Cost Estimates

In Chapter IV we described the combined cost measure that adds the costs for existing landfills and for replacement landfills into one present value spanning the existing landfill's remaining life plus the life of the new landfill that replaces it. The resulting annualized present value provided us with a single (levelized) cost for each landfill. These combined costs can be thought of as a weighted average of new and existing landfill costs. With a short remaining life, for example, existing costs will have a larger influence on combined costs than will new costs. Because the combined cost measure is sensitive to both new and existing costs and is simple to use, we will rely heavily on this measure of incremental costs in our analysis.

The national cost estimates for complying with the final rule differ depending on the length of the post-closure care period. Table VII-4 shows the annualized combined cost by the design to which the landfills were assigned. The total annualized costs range from $926 million for the 10-year care period to $989 million under the 40-year care scenario. As described in Chapter IV, these combined costs take account of existing state requirements for ground-water monitoring and containment and cover systems as appropriate. The majority of the total cost in either scenario results from landfills assigned to the uniform design standard. Even though only 27 percent of the landfills fall into this category, they account for roughly 60 percent of the total cost. The 56 percent of landfills that require no additional containment or cover systems under the performance standard account for between 18 and 20 percent of the total cost. Without corrective action, the total costs would vary between $926 million and $989 million, depending on the implementation decisions of different states.

Table VII-4

Final Rule: Annualized Combined Cost by Design; No Corrective Action

10- and 40-Year Post-Closure Care Periods

(dollars in millions)

Designs	**Percentage of Landfills**	**10-Year Post-Closure Annualized Cost**	**10-Year Post-Closure Percentage of Total Cost**	**40-Year Post-Closure Annualized Cost**	**40-Year Post-Closure Percentage of Total Cost**
Performance Standard					
Unlined/Vegetative Cover	56%	$165.3	17.9%	$193.9	19.6%
Unlined/Synthetic Cover	1	4.0	0.4	4.3	0.4
Synthetic Liner and Cover	16	179.1	19.3	189.2	19.1
Uniform Standard					
Composite Liner/Synthetic Cover (on-site clay)	20%	$483.6	52.2%	$502.7	50.9%
Composite Liner/Synthetic Cover (off-site clay)	7	94.2	10.2	98.7	10.0
Totals	100%	$926.2	100.0%	$988.8	100.0%

Corrective Action Costs

As described in Chapters IV, VI, and IX, corrective action imposes incremental costs on some landfills under the final rule, and for modeling purposes we based the incidence of corrective action costs on results from the risk model. Only existing units incur incremental costs for corrective action under the final rule since the design standards prevent releases from new units during the modeling period (i.e., the operating life and up to 40 years of post-closure care). Corrective action was triggered only at landfills in certain environmental settings and with certain designs, so the cost estimates are linked to unit size, net precipitation, and ground-water flow field. We were able to relate the cost results to surveyed landfills based on size and net precipitation only, since the surveyed units did not report detailed characteristics of the underlying aquifer.

As we described in Chapter IV, the approach for existing units is the same for all the regulatory options because existing units are assumed to have the same baseline design. The incidence and cost estimates, therefore, are the same for existing units under all five regulatory options. Landfills that trigger some response to contamination will fall into one of two categories: those with the most extensive contamination (i.e., those with plumes that exceed a risk threshold at the 600-meter well) will clean up plumes; those with less extensive contamination that trigger resource damage will replace the ground-water supply only and no cleanup costs are included. As with resource damage estimates, the cost to replace water supplies is greater if ground-water wells already exist near the landfill. If no wells are nearby, the option value cost is used.

Table VII-5 shows the fraction triggering corrective action and the cleanup cost, the fraction triggering replacement, and the relative costs of replacement depending on whether or not wells are downgradient from the unit. Overall, 9 percent of the existing units trigger corrective action and are assigned costs to clean up the plumes immediately. The costs per ton for cleanup range from $1.97 to $47.83 and the present values of the cleanup costs range from $4.1 million to $15.4 million with a median present value of $12.8 million. These are the most expensive cleanup costs estimated for this analysis because the plumes have been allowed to grow undetected from the existing units until the rule is in place and exceed trigger levels beyond 600 meters of plume length.

Table VII-5

All Options: Incidence of Corrective Action Costs for Existing Landfills

Size (TPD)	Net Precipitation (Inches per year)	Percentage of of Landfills In Settings That Incur Cleanup Costs	Weighted Annualized Cost per Ton	Percentage of Landfills In Settings That Will Replace Ground-water Supply	Weighted Annualized Cost per Ton to Replace: Wells In Place	No Wells
1-30	<0	0.0%	-	34.6%	$5.61	$0.80
	0-5	0.0	-	33.9	12.53	1.04
	5-15	0.0	-	79.6	20.21	1.02
	>15	2.3	$47.83	93.3	19.48	0.71
30-500	<0	0.0%	-	63.4%	$0.76	$0.07
	0-5	0.0	-	63.8	1.09	0.08
	5-15	43.3	$7.60	56.7	1.72	0.08
	>15	51.3	9.06	48.7	2.13	0.08
500+	<0	0.0%	-	34.6%	$0.20	$0.02
	0-5	0.0	-	79.5	0.38	0.02
	5-15	46.7	$1.97	53.3	0.53	0.02
	>15	72.8	2.37	27.2	0.63	0.02

The remaining columns in the table describe the incidence and costs for units where releases cause resource damage; the costs are for ground-water supply replacement when either current or future wells are affected. Approximately 58 percent of the landfills will trigger resource damage and be assigned replacement costs. As with the entire sample of landfills, the majority of these (54 percent) are assigned option value costs (they will replace the ground-water supply at some point in the future) because there are no wells downgradient. The remainder will replace the ground-water supply immediately.

In several scenarios (e.g., 30 to 500 TPD and more than 15 inches of net precipitation), cleanup or replacement costs are incurred at every landfill. Overall, however, 58 percent of the MSWLFs incur some current or future replacement cost. Including both corrective action (cleanup) and replacement, we estimated that 66 percent of the existing MSWLFs will trigger some response to releases. This estimate will remain the same for all the options analyzed below. These costs assume baseline (unlined/vegetative cover) units and therefore do not include incremental improvements in performance due to better covers if they are used.

Total Combined Costs Including Corrective Action

The total combined cost estimates are derived from the combined cost estimates in Table VII-4, which include costs for design and operating standards, and from the corrective action costs for existing units. Again the costs are presented as a range to reflect the different post-closure care periods that states may select. The total annualized combined cost of the final rule ranges between $975 million and $1,038 million, as shown on Table VII-6. The table disaggregates the costs by the design options described earlier. The distribution of costs across designs is quite similar to the distribution before corrective action was added. Overall, corrective action increases the compliance costs by 4.9 to 5.3 percent, depending on the length of the post-closure care period.

If the federal requirements that overlap with state regulations were not excluded from this national total, the annualized, combined cost would increase by approximately $450 million. A baseline that includes these state regulations would total $2.6 billion, as estimated by the model. The final rule would therefore represent an increase of approximately 40 percent above this baseline.

Table VII-6

Final Rule: Annualized Combined Cost by Design; Including Corrective Action

10- and 40-Year Post-Closure Care Periods

(dollars in millions)

Designs	Percentage of Landfills	10-Year Post-Closure Annualized Cost	10-Year Post-Closure Percentage of Total Cost	40-Year Post-Closure Annualized Cost	40-Year Post-Closure Percentage of Total Cost
Performance Standard					
Unlined/Vegetative Cover	56%	$170.3	17.5%	$198.9	19.1%
Unlined/Synthetic Cover	1	4.7	0.5	5.0	0.5
Synthetic Liner and Cover	16	195.1	20.0	205.2	19.8
Uniform Standard					
Composite Liner/Synthetic Cover (on-site clay)	20%	$506.6	52.0%	$525.7	50.7%
Composite Liner/Synthetic Cover (off-site clay)	7	98.3	10.0	102.8	9.9
Totals	100%	$975.0	100.0%	$1,037.6	100.0%

Another approach for presenting the results is to disaggregate the total costs by landfill size. Table VII-7 confirms the inverse relationship between costs per ton (which are highest for the more numerous small landfills) and total combined costs. The largest landfills (i.e., those that manage more than 1,125 TPD) account for only 2.6 percent of MSWLFs, yet these units bear roughly 20 percent of the total cost of the rule. Half of the incremental cost of the rule is attributable to only the largest 11 percent of the landfills in the United States. The table also indicates that the cost burden shifts slightly toward smaller units when the post-closure period is lengthened from 10 to 40 years. Since ground-water monitoring costs are much more significant for small landfills, the extra years of monitoring affect their compliance costs more.

Table VII-7

Final Rule: Annualized Combined Cost by Size; Including Corrective Action

10- and 40-Year Post-Closure Care Periods

(dollars in millions)

		10-Year Post-Closure		40-Year Post-Closure	
Size (TPD)	**Percentage of MSWLFs**	**Annualized Cost**	**Percentage of Total Cost**	**Annualized Cost**	**Percentage of Total Cost**
10	51.3%	$62.0	6.4%	$71.7	6.9%
25	17.0	107.7	11.0	118.6	11.4
75	13.1	160.8	16.5	171.6	16.6
175	7.3	149.2	15.3	156.8	15.1
375	5.5	146.9	15.1	154.7	14.9
750	3.1	154.0	15.8	159.7	15.4
1,500	2.6	194.4	19.9	204.5	19.7
	100.0%[a]	$975.0	100.0%	$1,037.6	100.0%

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[a] Does not add because of rounding.

The median cost per year for landfills under the final rule ranges between $44,400, assuming 10 years of post-closure care, and $53,700, assuming 40 years. The median incremental cost per ton ranges between $11.61 and $13.82 depending on the length of the post-closure care period. The lowest cost is $0.58 per ton (for a 4,400-TPD landfill) and the highest is $52.81 (at a 10-TPD landfill); if corrective action costs are excluded from the analysis, the incremental costs range from $0.51 to $48.02 per ton. Table VII-8 shows the distribution of incremental costs. The longer post-closure care period shifts incremental costs per ton up across all categories; for example, the share of landfills with costs above $25 per ton increases from 15.0 percent to 17.1 percent with the additional 30 years of monitoring.

Table VII-8

Final Rule: Distribution of Incremental Cost per Ton; Including Corrective Action

Incremental Cost per Ton	10-Year Post-Closure	40-Year Post-Closure
$0 – $1	3.3%	2.4%
$1 – $10	39.1	28.6
$10 – $25	42.6	51.9
$25 – $50	15.0	16.7
$50 – $100	0.0	0.4
	100.0%	100.0%

We also compared the incremental costs under the final rule to the level of current annual operating costs for landfills as reported in the survey. Because we suspect that these reported costs are underestimated (see Figure VII-1), these percentage increases are somewhat inflated, but they provide perspective on the magnitude of the increases. Assuming a 40-year post-closure care period, the final rule will at least double the annual operating expenses at just over 50 percent of the landfills. Although counter to our expectations, many large percentage increases occur at large landfills as well as at small ones. We attribute this to inconsistencies in the reporting of operating costs in the survey. While we are adding O&M costs and amortized capital costs, the survey responses may only include O&M costs; therefore the cost increases above reported costs may be less than indicated by this measure.

Alternative 1: 1988 Proposal

The proposal imposes different costs on MSWLFs, depending on the design necessary to comply with the state-specified design goal. First, we discuss unit cost estimates for new and existing landfills. Second, we aggregate the unit costs to calculate national cost estimates.

The *unit* costs presented assume full compliance with the proposal and do not take into account states that have some of these requirements already in place. However, the recognition of current state requirements *is* reflected in the computation of combined costs and aggregate national costs.

Unit Costs for New Landfills

Table VII-9 presents the annualized incremental costs per ton for each of the design options we modeled for the proposal. These unit costs assume that landfills have a 20-year operating life and that construction begins immediately. As described in Chapter IV, the more stringent designs are assigned to units at which the less elaborate designs are ineffective at reducing risk below 10^{-5}. The largest incremental cost is imposed by the synthetic cover, synthetic liner, and leachate collection system installed on the high-risk units: the incremental costs per ton are two to six times higher for the synthetic/synthetic design than for the unlined/vegetative design.

Table VII-9

Alternative 1: Incremental Costs for New Landfills
40-Year Post-Closure Period

(dollars per ton)

Size (TPD)	Unlined/ Vegetative Cover	Unlined/ Synthetic Cover	Synthetic Liner/ Synthetic Cover
10	$22.30	$26.47	$43.41
25	12.95	16.38	30.07
75	6.03	8.35	17.37
175	3.98	5.83	12.94
375	2.47	3.89	9.30
750	1.45	2.46	6.26
1,500	1.08	2.20	6.43

The costs for all designs include the general facility standards described in Chapter IV and the cost to assess compliance with the design goal. For the unlined/vegetative design, these costs account for about 15 percent of the incremental cost at the 10-TPD landfills and 40 percent of the cost at the 1,500-TPD units. These costs include gas monitoring, run-on and run-off controls, and higher engineering and quality assurance (QA) fees on excavation and design. The remainder (85 percent for the 10-TPD units and 60 percent for the 1,500-TPD units) is attributable to ground-water monitoring including capital costs for wells, a hydrogeologic study to characterize ground-water flow and to determine appropriate placement for wells, and monitoring through the 40-year post-closure care period. The costs for monitoring are higher than they are under the final rule because costs per sample are $644 instead of $360 for the final rule.

The incremental costs for the other two modeled designs are influenced substantially by the cover and containment system costs. The synthetic cover system adds 19 percent to the incremental costs for the 10-TPD unit and more than doubles the incremental costs for the 1,500-TPD landfills. As discussed above, the cover costs do not exhibit large returns to scale because of the depth limitation imposed on units in the analysis. Overall, the 10-TPD incremental cost is 16 percent cover cost, 72 percent ground-water monitoring cost, and 12 percent general facility standards. Given the weaker returns to scale for cover costs, they dominate the incremental costs for the 1,500-TPD MSWLFs, accounting for 50 percent of the incremental cost. Ground-water monitoring contributes another 30 percent to the incremental costs.

The containment system, where required, is the largest component of the costs for the larger landfills. The liner, LCS, and cost of leachate treatment account for 66 percent of the incremental costs for the 1,500-TPD landfills but only 39 percent of the costs for the 10-TPD landfills. At the largest units, the cover accounts for 17 percent and monitoring adds another 10 percent. For 10-TPD units, the corresponding components add 10 and 44 percent, respectively. Monitoring costs hit the smaller landfills particularly hard because of the relatively high fixed cost component.

If the post-closure period were shortened to 30 years (the minimum allowed under the proposal), the costs of post-closure care and monitoring would fall. Table VII-10 shows the difference in annualized cost per ton for each size category between 30 and 40 years of post-closure care. The largest effect of this change would be on 10-TPD landfills assigned to the unlined/vegetative design; a shift to a 30-year post-closure care period would cut costs by 5 percent at this unit ($1.08 divided by $22.30 from Table VII-9).

Table VII-10

Alternative 1: Incremental Cost Between 30 and 40 Years of Post-Closure Care for New Landfills

(dollars per ton)

Size (TPD)	Difference In Cost per Ton
10	- $1.08
25	- 0.60
75	- 0.23
175	- 0.14
375	- 0.08
750	- 0.05
1,500	- 0.04

Unit Costs for Existing Landfills

As in our analysis of the final rule, we fixed the remaining life at 10 years for the purpose of comparing incremental compliance costs; elsewhere in the analysis we used the facilities' actual remaining lives to estimate costs. The requirements for existing units are the same as they are for the final rule. For landfills with negative net precipitation, the incremental costs reflect ground-water monitoring and general facility standards. As with new units, the monitoring well costs are only applied in states without equivalent monitoring programs, but this adjustment is not reflected in the unit costs presented here. Table VII-11 lists the incremental cost for these unlined/vegetative designs. Assuming a 40-year post-closure care period, costs at the 10-TPD units are primarily from ground-water monitoring costs (86 percent), with the balance due to general facility standards for gas monitoring, run-on and run-off controls, and higher fees. Ground-water monitoring accounts for 56 percent of the incremental cost for the 1,500-TPD units. Monitoring is a more significant share of existing costs under the proposal than it is for the final rule because sampling costs are higher under the proposal.

Table VII-11

Alternative 1: Incremental Costs for Existing Landfills

30- and 40-Year Post-Closure Care Periods

(dollars per ton)

	30-Year Post-Closure		40-Year Post-Closure	
Size (TPD)	**Unlined/ Vegetative Cover**	**Unlined/ Synthetic Cover**	**Unlined/ Vegetative Cover**	**Unlined/ Synthetic Cover**
10	$34.39	$44.18	$36.93	$46.71
25	20.29	28.32	21.70	29.73
75	9.72	15.17	10.27	15.72
175	6.53	10.86	6.86	11.19
375	4.11	7.44	4.30	7.62
750	2.44	4.79	2.55	4.90
1,500	1.85	4.46	1.93	4.54

Those units that will not meet the "minimize leachate formation" performance standard with a vegetative cover were modeled with a synthetic cover at closure at higher costs. These incremental costs are also presented in Table VII-11. Compared with units with a 40-year post-closure care period and the unlined/vegetative design, these landfills incur costs between 26 percent and 135 percent higher. All the increase is due to the cost of the cover system. For the smallest units, the cover costs contribute 21 percent to incremental costs and monitoring adds 68 percent. At the 1,500-TPD MSWLFs, cover costs account for 58 percent of the costs, with ground-water monitoring the next most expensive component at 24 percent.

If the post-closure care period is set at 30 years, the unit costs shift down by, at most, 7 percent. Because most of the savings results from ground-water monitoring during post-closure care, the unit costs include a lower share of monitoring costs, while cover costs and general facility standards are slightly more significant.

Combined Costs and National Cost Estimates

With the post-closure care period set at 30 years, the annualized combined cost for all landfills is $893 million, excluding costs for corrective action, which we estimate from risk model results. With a 40-year post-closure care period, the annualized combined cost is $918 million, an increase of less than 3 percent. Though they account for only 28 percent of the units, the units with synthetic liners and covers incur 63 percent of total compliance costs.

As described in Chapter IV, corrective action is only triggered in some environmental settings and for those MSWLFs that, despite their synthetic liner and cover, will not meet the design goal. Given the results of the risk model, no new units trigger corrective action; existing units do incur costs for corrective action just as they do under the final rule. The total combined cost estimates are derived from the combined cost estimates given above, which include general facility standards and design and operating standards for new and existing landfills, and from the corrective action costs for existing units. The total annualized combined cost assuming 30 years of post-closure care is $941.7 million. The existing corrective action costs added 5 percent to the combined cost reported for the design standards. If post-closure care extends to 40 years, the total annualized combined cost is $966.6 million, an increase of 2.6 percent as shown on Table VII-12. The 28 percent of landfills with the synthetic/synthetic design incur 64 percent of the costs and, overall, the addition of 10 years to post-closure care adds less than 3 percent to the total costs.

Table VII-12

Alternative 1: Annualized Combined Cost by Design; Including Corrective Action

30- and 40-Year Post-Closure Care Periods

(dollars in millions)

Design	Percentage of Landfills	30-Year Post-Closure Annualized Cost	30-Year Post-Closure Percentage of Total Cost	40-Year Post-Closure Annualized Cost	40-Year Post-Closure Percentage of Total Cost
Unlined/Vegetative Cover	61.0%	$243.5	25.9%	$255.1	26.4%
Unlined/Synthetic Cover	11.0	93.5	9.9	96.2	10.0
Synthetic Liner and Cover	28.0	604.7	64.2	615.3	63.6
Totals	100.0%	$941.7	100.0%	$966.6	100.0%

Table VII-13 summarizes national costs by size category. Given the relatively minor change in total costs resulting from the change in the post-closure care period, the cost distribution is similar between the two scenarios. The smallest two-thirds of the landfills (less than 50 TPD) account for just over 20 percent of the total costs.

Table VII-13

Alternative 1: Annualized Combined Cost by Size; Including Corrective Action

30- and 40-Year Post-Closure Care Periods

(dollars in millions)

		30-Year Post-Closure		**40-Year Post-Closure**	
Size (TPD)	**Percentage of Landfills**	**Annualized Cost**	**Percentage of Total Cost**	**Annualized Cost**	**Percentage of Total Cost**
10	51.3%	$77.7	8.3%	$81.7	8.3%
25	17.0	115.0	12.2	119.5	12.4
75	13.1	150.8	16.0	155.2	16.1
175	7.3	134.8	14.3	137.9	14.3
375	5.5	145.4	15.4	148.5	15.4
750	3.1	133.0	14.1	135.1	14.0
1,500	2.6	185.0	19.7	188.7	19.5
	100.0%[a]	$941.7	100.0%	$966.6	100.0%

[a] Does not add because of rounding.

Typical facility costs (medians) are slightly higher under this alternative than they are under the final rule. The medians range from $67,500 (30-year post-closure) to $71,600 (40-year post-closure). The higher cost compared to the final rule results from the higher monitoring costs imposed on MSWLFs by the proposal. Median costs per ton range from $16.74 to $17.82, also higher than under the final rule. Table VII-14 presents the distribution of incremental costs per ton. The proposal never imposes incremental costs per ton above $50, but the table indicates that typical costs are higher under the proposal than they are for the final rule. Assuming a 40-year post-closure period, the proposal results in costs above $25 per ton for 25.9 percent of the landfills compared with 17.1 percent under the final rule.

Table VII-14

Alternative 1: Distribution of Incremental Cost per Ton; Including Corrective Action

Incremental Cost per Ton	30-Year Post-Closure	40-Year Post-Closure
$0 - $1	1.2%	1.1%
$1 - $10	29.0	28.4
$10 - $25	47.7	44.6
$25 - $50	22.1	25.9
$50 - $100	0.0	0.0
	100.0%	100.0%

Alternative 2: Subtitle C Design Standards

This alternative imposes uniform requirements on all landfills, regardless of their location. The requirements are more stringent than those modeled for the final rule: the ground-water monitoring parameters modeled are more extensive (although still less than those required by the proposal) and the cover and containment system designs comply with those required at Subtitle C landfills. We did exclude capital costs for wells from the incremental cost for landfills in states with adequate existing requirements, and landfills in four states received at least some credit for existing state requirements for either covers or containment systems. Finally, the post-closure care period is varied between 10 and 40 years as it is under the final rule.

New Landfills

The incremental costs per ton for new units under Alternative 2 are presented in Table VII-15. In order to assign the correct costs, we used MSWLF survey results to determine whether the clay needed for the landfill cover and liner was available at the site: the 76.7 percent of units reporting clay or sandy clay as the primary or secondary soil type are assumed to have adequate clay on site.

Table VII-15

Alternative 2: Incremental Costs for New Landfills

10- and 40-Year Post-Closure Care Periods

(dollars per ton)

Size (TPD)	10-Year Post-Closure On-Site Clay	10-Year Post-Closure Off-Site Clay	40-Year Post-Closure On-Site Clay	40-Year Post-Closure Off-Site Clay
10	$61.46	$76.25	$65.79	$80.58
25	45.41	57.43	47.74	59.76
75	28.25	36.30	29.14	37.20
175	21.63	28.00	22.15	28.52
375	15.73	20.52	16.01	20.80
750	11.01	14.43	11.17	14.59
1,500	11.77	15.58	11.89	15.69

The first costs we will review are for those MSWLFs with on-site clay. The additional 30 years of post-closure care can add up to 7 percent to the incremental cost per ton. If we limit our review to the longer post-closure care example, the costs range from $65.79 per ton to $11.17 per ton. The majority of the cost is attributable to the cover and containment system requirements. The liner and cover costs account for an even greater percentage of the costs than they do for the uniform design under the final rule. In fact, the LCS, containment, and cover costs account for between 70 and 91 percent of the incremental costs.

The dominant component of the $65.79 for 10-TPD units is the containment system and the accompanying leachate collection and treatment, which account for about 56 percent of the incremental cost. The cover system accounts for about 14 percent of the increment while ground-water monitoring adds 25 percent. The balance includes the costs of the other requirements, such as run-on and run-off controls, gas monitoring, ground-water monitoring, and higher fees and QA.

For the 1,500-TPD facilities, the incremental cost is $11.89, of which 77 percent is for the containment system and leachate collection and treatment. Fifteen percent is for the cover system and the balance is made up of all other requirements. The ground-water monitoring costs account for 5 percent of the incremental cost per ton for the large units.

The unit cost for clay from off site is over twice the cost for on-site clay. The new-unit costs per ton for Alternative 2 are, accordingly, between 22 and 32 percent higher if off-site clay is used.

Existing Landfills

As with the final rule, we fixed the remaining life at 10 years to illustrate the magnitude of existing costs across size categories. The incremental costs per ton are lower for existing units (assuming 10 years of remaining life) than for new, even though the shorter amortization period might suggest that the costs should be higher. They are lower because containment and leachate management systems costs are not included for existing units. Because containment costs are lower, ground-water monitoring has a larger share of the costs and, therefore, the additional years of post-closure will increase costs more for existing units than for new units. Depending on the landfill size, the incremental costs of 40-year post-closure represents an increase of up to 24 percent compared with 10-year post-closure.

Table VII-16 presents the incremental costs per ton for all possible scenarios. For units with 40-year post-closure and on-site clay, the incremental costs per ton range from $52.37 for 10-TPD units to $5.93 for 1,500-TPD units. At small MSWLFs, 52 percent of the incremental costs is due to ground-water monitoring, which includes wells and monitoring. The cover system accounts for another 36 percent of the cost. The balance consists of post-closure costs, higher fees for construction, surface-water controls, and gas monitoring.

Table VII-16

Alternative 2: Incremental Costs for Existing Landfills

10- and 40-Year Post-Closure Care Periods

(dollars per ton)

Size (TPD)	10-Year Post-Closure		40-Year Post-Closure	
	On-Site Clay	Off-Site Clay	On-Site Clay	Off-Site Clay
10	$42.41	$50.75	$52.37	$60.91
25	28.43	35.53	33.90	41.00
75	16.57	21.44	18.67	23.53
175	12.27	16.15	13.47	17.36
375	8.48	11.42	9.14	12.09
750	5.76	7.89	6.14	8.27
1,500	5.66	8.02	5.93	8.29

For 1,500-TPD landfills, the cover costs dominate all others, accounting for 71 percent of the incremental costs. Another 16 percent is from ground-water monitoring.

Units that must obtain clay from off site have incremental costs between 16 and 42 percent higher than those using on-site clay. As expected, the increases are larger for large units, where cover costs dominate the incremental cost estimates. The incremental costs for 10-TPD units are dominated by ground-water monitoring (45 percent) and the cover system (45 percent). At 1,500-TPD landfills, 79 percent of the incremental cost is attributable to the cover system and 12 percent to monitoring.

Combined Costs and National Cost Estimates

Given the relatively high cost of the cover and containment systems and the fact that all the units are required to install these more elaborate systems, we expect this to be the most expensive option. The annualized combined costs (without corrective action) for the 6,034 landfills under Alternative 2 is $2,702 million with a 10-year post-closure assumption and $2,795 million if post-closure lasts 40 years.

As with the other options, we used the risk model to estimate the fraction of landfills that would trigger corrective action at new and existing units. Because of the design standards employed for new facilities, corrective action is never triggered for new units; existing landfills trigger corrective action according to the results shown in Table VII-5.

Table VII-17 presents the annualized combined cost including corrective action for existing units. The remedial measures add 3 percent to the total of the design and operating costs, a small percentage increase because of the magnitude of the design costs already included. The total combined cost of about $2.8 billion is the highest of any option analyzed and is dominated by very high costs for design standards at the largest units. If credits for overlapping state programs were not taken into account, the annualized combined costs would increase by about $150 million.

Clay is available on-site at 76.7 percent of the landfills. The survey data also show that the typical size of units with clay on-site is slightly larger than those without clay. As a result of the relationship between size and clay availability, the mean cost per unit with on-site clay is higher than the mean without clay, even though off-site clay is much more expensive. Overall, units with on-site clay account for 83 percent of the cost.

Table VII-17

Alternative 2: Annualized Combined Cost by Soil Type; Including Corrective Action

10- and 40-Year Post-Closure Care Periods

(dollars in millions)

		10-Year Post-Closure		40-Year Post-Closure	
Soil Type	**Percentage of Landfills**	**Annualized Cost**	**Percentage of Total Cost**	**Annualized Cost**	**Percentage of Total Cost**
Clay On-Site	76.7%	$2,279.2	82.8%	$2,355.9	82.9%
Clay Off-Site	23.3	472.0	17.2	487.6	17.1
Total	100.0%	$2,751.2	100.0%	$2,843.5	100.0%

Table VII-18 summarizes the costs of Alternative 2 by landfill size. This alternative shifts a greater share of total costs to the largest landfills than any other option; it is the most capital-intensive option and even though scale economies are significant, they break down somewhat at the largest landfills. The table shows that the largest 2.6 percent of the units bear more than one-fourth of the cost. Also, because of the high costs for design components, the additional years of monitoring under the 40-year scenario add only 3 percent to the total cost.

Table VII-18

Alternative 2: Annualized Combined Cost by Size; Including Corrective Action

10- and 40-Year Post-Closure Care Periods

(dollars in millions)

		10-Year Post-Closure		40-Year Post-Closure	
Size (TPD)	**Percentage of MSWLFs**	**Annualized Cost**	**Percentage of Total Cost**	**Annualized Cost**	**Percentage of Total Cost**
10	51.3%	$175.3	6.4%	$191.0	6.7%
25	17.0	279.1	10.1	296.5	10.4
75	13.1	411.2	14.9	427.9	15.1
175	7.3	368.3	13.4	379.6	13.4
375	5.5	444.8	16.2	455.9	16.0
750	3.1	365.5	13.3	373.0	13.1
1,500	2.6	707.0	25.7	719.6	25.3
	100.0%[a]	$2,751.2	100.0%	$2,843.5	100.0%

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[a] Does not add because of rounding.

Median costs per facility per year range from $163,000 to $177,000, depending on the length of the post-closure care period. Median costs per ton are much higher under this alternative as well. Table VII-19 shows that incremental costs above $50 per ton are common under this alternative while virtually no landfills faced costs this high for the final rule or the proposal. Median costs per ton for the two post-closure scenarios are $39.64 and $41.50, each roughly triple the corresponding medians under the final rule.

Table VII-19

Alternative 2: Distribution of Incremental Cost per Ton; Including Corrective Action

Incremental Cost per Ton	10-Year Post-Closure	40-Year Post-Closure
\$0 – \$1	0.6%	0.2%
\$1 – \$10	6.0	5.4
\$10 – \$25	26.4	25.4
\$25 – \$50	27.2	26.9
\$50 – \$100	39.8	42.1
	100.0%	100.0%

Alternative 3: Categorical Design Standards

Alternative 3 imposes different costs on MSWLFs depending on their location and requires the use of engineered controls to meet performance standards for infiltration and leachate release. We will present unit costs for new and existing landfills and aggregate them to produce national combined cost estimates for the alternative.

New Landfills

The costs for new units assume that construction begins immediately and that the units will operate for 20 years. In the economic analysis and for the combined costs, the starting date is delayed until the expected life of the existing unit is reached. Table VII-20 summarizes the annualized incremental costs per ton for each design category, described in Chapter IV. The costs for a synthetic/synthetic design with LCS are the highest, between 2.3 and 6.9 times higher than the incremental costs for the least expensive design, depending on size.

Table VII-20

Alternative 3: Incremental Costs for New Landfills

40-Year Post-Closure Care

(dollars per ton)

Size (TPD)	Unlined/ Vegetative Cover	Unlined/ Synthetic Cover	Clay (on-site)/ Vegetative Cover	Clay (off-site)/ Vegetative Cover
10	$15.80	$19.97	$23.42	$29.97
25	9.36	12.79	15.61	20.97
75	4.67	6.99	8.87	12.49
175	3.19	5.03	6.52	9.39
375	2.03	3.45	4.59	6.78
750	1.20	2.21	3.00	4.54
1,500	0.91	2.02	2.92	4.64

Size (TPD)	Synthetic Liner/ Vegetative Cover & LCS	Synthetic Liner/ Synthetic Cover & LCS	Unlined/ Vegetative Cover & LCS	Unlined/ Synthetic Cover & LCS
10	$32.91	$36.91	$28.68	$32.68
25	23.20	26.48	19.75	23.03
75	13.78	16.00	11.46	13.69
175	10.38	12.15	8.55	10.31
375	7.50	8.86	6.10	7.46
750	5.05	6.01	4.06	5.02
1,500	5.19	6.25	4.09	5.15

The costs for all designs included the same general facility standards imposed under the final rule. We also included a fixed cost for determining precipitation ($200 for new and existing units) and travel time ($3,000 for new units only). The incremental cost per ton for the nondesign requirements ranges from $15.80 for the smallest unit down to $0.91 for the largest. About 80 percent of this incremental cost for the 10-TPD units arises from ground-water monitoring. The costs include capital costs for wells, the hydrogeologic study to characterize ground-water flow and determine appropriate placement for wells, and the semi-annual monitoring cost through post-closure. Remaining costs include gas monitoring, run-on and run-off controls, and additional fees for design quality assurance. Ground-water monitoring accounts for only 52 percent of the incremental cost for the largest units. The dominant remaining costs are the higher engineering fees associated with excavation and site-development costs and quality assurance fees for the design. These are the only costs for units assigned to the unlined/vegetative cover design.

The costs for the remaining designs are higher because of the various combinations of covers, liners, and leachate collection systems.

The next design shown in Table VII-20 is unlined with a synthetic cover, a design that was also analyzed under the final rule. The synthetic cover cost adds about $4 per ton to the cost of the 10-TPD unlined/vegetative (U/V) design. The $19.97 incremental cost for 10-TPD units is dominated by ground-water monitoring (63 percent) and the cover (21 percent). For 1,500-TPD landfills, the cover dominates incremental costs, accounting for 55 percent of the cost, while ground-water monitoring adds another 23 percent.

The designs with clay liners are applicable when the owner or operator must increase the travel time but no LCS is required. The clay liner and ground-water monitoring are, by far, the most expensive components of the costs for these designs. For the smallest model size the liner accounts for 33 percent of the incremental cost if clay is available on-site, or 47 percent if clay is brought from off-site. Ground-water monitoring accounts for 54 percent of the on-site clay design cost and 42 percent of the off-site clay cost. For 1,500-TPD units, the liner is even more dominant, accounting for 69 percent of the cost using on-site clay and 80 percent if off-site clay is used.

The next design, synthetic liner with LCS, is more expensive than either of the clay designs. The combined liner/LCS design adds over $17 per ton to the cost for a 10-TPD unlined design. The containment and leachate system together account for 52 percent of the incremental design cost for 10-TPD units. For 1,500-TPD units, the system accounts for 83 percent of the cost.

The most expensive design analyzed is the synthetic liner, synthetic cover, and LCS combination. This design is also analyzed under the final rule. For the largest units, the cover, liner, and LCS account for 85 percent of the incremental cost: of these components the LCS is the most expensive, accounting for about half of the increment. At the 10-TPD size, the three requirements account for 58 percent of incremental costs; ground-water monitoring adds 34 percent.

The final two designs analyzed for new units add components of other designs to the unlined/vegetative cover design. Each of the designs has an LCS over native soil and is only assigned if the MSWLF is located in clay. The cost per ton for the unlined design with an LCS is $28.68 for the 10-TPD and ranges down to $4.06 for the 750-TPD. The LCS dominates the incremental cost estimate, accounting for 44 percent of the 10-TPD cost and 78 percent of the 1,500-TPD cost. The final design adds a synthetic cover to the LCS over native clay. The LCS cost is still the largest component at 10-TPD landfills, accounting for 39 percent, followed by ground-water monitoring (38 percent), and the cover (13 percent). At 1,500-TPD units the LCS accounts for 62 percent of the incremental cost, the cover 22 percent, and ground-water monitoring only 9 percent.

The costs in Table VII-20 assume a 40-year post-closure care period. If states allowed a 10-year period instead, the costs for all of the designs would be reduced by between 1 and 17 percent. The difference is shown in Table VII-2; in each case, the share of the incremental costs attributable to ground-water monitoring would decrease.

Existing Landfills

The requirements for existing units are the same as for the new unlined/vegetative cover design if units are in areas with zero or negative net precipitation. If net precipitation is positive, the unlined/synthetic design is required. Again, we emphasize that these designs were chosen for modeling purposes and assumed to meet the performance standards. Table VII-21 presents incremental costs per ton for each design assuming 10 years of remaining life. As with new units, the largest contributor to incremental cost for the 10-TPD units is ground-water monitoring, which accounts for 75 to 80 percent of the unlined/vegetative cost, depending on the length of the post-closure care period. Monitoring is also the dominant cost for the unlined/synthetic cover design. Depending on post-closure, monitoring contributes 51 to 58 percent of the incremental cost. The synthetic cover accounts for 27 to 32 percent of the unlined/synthetic design. If we look at the component costs for the largest units, the single most expensive part of the unlined/vegetative cover design is ground-water monitoring (44 to 48 percent). The unlined/synthetic cover design is 61 to 64 percent cover cost and 16 to 19 percent ground-water monitoring. Gas monitoring, quality-assurance fees, and run-on and run-off controls make up the balance.

We have shown for both new and existing units that ground-water monitoring dominates the cost estimates for small units with no location-related cover or liner requirements. Once the units are subject to these site-specific requirements, however, the cover or containment system cost dominates the incremental cost estimates.

Table VII-21

Alternative 3: Incremental Costs for Existing Landfills

10- and 40-Year Post-Closure Care Periods

(dollars per ton)

Size (TPD)	10-Year Post-Closure Unlined/ Vegetative Cover	10-Year Post-Closure Unlined/ Synthetic Cover	40-Year Post-Closure Unlined/ Vegetative Cover	40-Year Post-Closure Unlined/ Synthetic Cover
10	$20.57	$30.35	$26.75	$36.53
25	12.57	20.60	16.00	24.03
75	6.75	12.20	8.10	13.55
175	4.77	9.10	5.58	9.91
375	3.12	6.45	3.59	6.92
750	1.85	4.21	2.14	4.50
1,500	1.43	4.04	1.65	4.25

Combined Costs and National Cost Estimates

For this alternative, the annualized combined cost for all landfills ranges between $1,121 million and $1,183 million, depending on the length of the post-closure care period. These costs exclude corrective action, however. Corrective action costs were provided from the risk model results. As with the final rule and Alternatives 1 and 2, the risk model indicated that given the designs to which new landfills are assigned, none of them will trigger corrective action within the operating plus post-closure care period. Corrective action costs were assigned to existing units as described under the final rule (see Table VII-5). We estimated that 66 percent of the existing units trigger some response to releases.

Table VII-22 shows the Alternative 3 costs including corrective action; corrective action does not have a large effect on total costs because more is spent on design standards than under the final rule. The landfills assigned to the synthetic/synthetic design account for about 80 percent of costs. The total cost of this option is 19 to 20 percent more expensive than the final rule, depending on the length of the post-closure care period.

Table VII-22

Alternative 3: Annualized Combined Cost by Design; Including Corrective Action
10- and 40-Year Post-Closure Care Periods

(dollars in millions)

Design	Percentage of Landfills	10-Year Post-Closure Annualized Cost	10-Year Post-Closure Percentage of Total Cost	40-Year Post-Closure Annualized Cost	40-Year Post-Closure Percentage of Total Cost
Unlined/Vegetative	17.6%	$49.5	4.2%	$58.9	4.8%
Unlined/Synthetic	2.1	25.2	2.2	27.4	2.2
Clay/Vegetative	5.3	45.6	3.9	49.1	4.0
Synthetic/Vegetative and LCS	7.8	23.9	2.0	26.4	2.2
Synthetic/Synthetic and LCS	56.1	938.0	80.2	977.3	79.3
Unlined/Vegetative and LCS	3.5	8.0	0.7	9.0	0.7
Unlined/Synthetic and LCS	7.6	79.3	6.8	84.0	6.8
Total	100.0%	$1,169.5	100.0%	$1,232.1	100.0%

The distribution of costs by landfill size is shown on Table VII-23. The additional 30 years of post-closure care add 5 percent to the total cost and also shift costs slightly toward smaller landfills. The 68 percent of landfills smaller than 50 TPD bear 18 percent of the cost with a 40-year post-closure period, but 17.3 percent if post-closure lasts only 10 years. Median costs per facility are $59,200 with 10 years of post-closure care and $68,500 if it is extended to 40 years.

Table VII-23

Alternative 3: Annualized Combined Cost by Size; Including Corrective Action

10- and 40-Year Post-Closure Care Periods

(dollars in millions)

		10-Year Post-Closure		40-Year Post-Closure	
Size (TPD)	**Percentage of MSWLFS**	**Annualized Cost**	**Percentage of Total Cost**	**Annualized Cost**	**Percentage of Total Cost**
10	51.3%	$76.9	6.6%	$86.6	7.0%
25	17.0	125.2	10.7	136.2	11.0
75	13.1	174.8	14.9	185.6	15.1
175	7.3	168.1	14.4	175.7	14.3
375	5.5	192.7	16.5	200.5	16.3
750	3.1	164.2	14.0	169.8	13.8
1,500	2.6	267.6	22.9	277.7	22.5
Total	100.0%[a]	$1,169.5	100.0%	$1,232.1	100.0%

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[a] Does not add because of rounding.

Median costs per ton range between $14.92 and $17.03 depending on the post-closure care period. These costs fall between the median costs for the final rule (which are lower) and for the proposal. Table VII-24 summarizes the distribution of incremental costs per ton under Alternative 3. Median costs never exceed $50 per ton (the maximum is $48.05 assuming 40 years of post-closure care), but the share of costs above $25 is higher than it is for either the final rule or the proposal.

Table VII-24

Alternative 3: Distribution of Incremental Cost per Ton; Including Corrective Action

Incremental Cost per Ton	10-Year Post-Closure	40-Year Post-Closure
$0 - $1	2.2%	1.5%
$1 - $10	30.2	24.4
$10 - $25	38.0	40.7
$25 - $50	29.6	33.4
$50 - $100	0.0	0.0
	100.0%	100.0%

Alternative 4: Statutory Minimum

As the name implies, this option requires similar provisions to those mandated in the 1984 Hazardous and Solid Waste Amendments. All units monitor ground water through the post-closure care period and perform corrective action as required; no location standards or general facility standards are included. As with Alternative 2, the requirements are imposed uniformly on all landfills, regardless of their location. Since the alternative includes no containment system or leachate collection system requirements, the requirements are the same for existing units and new units. As in the other options, capital costs for ground-water monitoring wells are not included for landfills in states with similar programs.

New Landfills

As in the other options, new landfills are assumed to begin operation immediately and receive waste for 20 years. Given these assumptions, the incremental costs per ton are summarized in Table VII-25. Among new units, the costs range from $12.33 for the 10-TPD landfills with a 40-year post-closure care period to $0.39 for the 1,500-TPD units and 10-years of post-closure care. The costs are due entirely to ground-water monitoring. The additional years of post-closure add 14 to 24 percent to the 10-year costs, depending on the size of the landfill.

Table VII-25

Alternative 4: Incremental Costs for New Landfills
10- and 40-Year Post-Closure Care Periods

(dollars per ton)

Size (TPD)	10-Year Post-Closure	40-Year Post-Closure
10	$9.95	$12.33
25	5.98	7.31
75	2.87	3.38
175	1.98	2.28
375	1.24	1.41
750	0.66	0.75
1,500	0.39	0.45

Existing Landfills

With the remaining life set at 10 years, the incremental costs for existing units are higher than for new units. They range from $20.71 per ton to $0.61 per ton and are listed in Table VII-26. The same requirements are imposed on both new and existing units, but the existing MSWLFs have only half the remaining life over which to amortize capital costs. Monitoring accounts for the entire compliance cost without corrective action.

Table VII-26

Alternative 4: Incremental Costs for Existing Landfills
10- and 40-Year Post-Closure Care Periods

(dollars per ton)

Size (TPD)	10-Year Post-Closure	40-Year Post-Closure
10	$15.14	$20.71
25	9.18	12.30
75	4.54	5.73
175	3.17	3.87
375	2.01	2.40
750	1.06	1.28
1,500	0.61	0.77

Combined Costs and National Cost Estimates

The combined annualized cost is between $178 million and $230 million, depending on post-closure care. These costs do not include corrective action costs, which are presented later. These costs assume sampling costs that are the same ($360) as those described for the final rule. We also excluded costs for ground-water monitoring wells at landfills in states with similar monitoring well requirements.

Corrective action costs more than triple the cost of the alternative, adding approximately $524 million to the combined annualized cost of the alternative. Table VII-27 summarizes the incidence of corrective action costs at new units. Overall, the risk model reports that 32.9 percent of new units trigger corrective action within 10 years of closure, and 34.8 percent trigger within 40 years of closure. The results also indicate that the later corrective actions are slightly less expensive (i.e., the costs are discounted more heavily) so the average corrective action costs are slightly lower with a longer post-closure care period.

Table VII-27

Alternative 4: Incidence of Corrective Action Costs for New Landfills

10- and 40-Year Post-Closure Care Periods

		10-Year Post-Closure		40-Year Post-Closure	
Size (TPD)	**Net Precipitation (inches per year)**	**Percentage of Landfills in Settings that Incur Corrective Action Costs**	**Weighted Annualized Costs per Ton**	**Percentage of Landfills in Settings that Incur Corrective Action Costs**	**Weighted Annualized Costs per Ton**
1-30	<0	0.0%	--	0.0%	--
	0-5	0.0	--	0.0	--
	5-15	51.7	$54.38	56.3	$53.72
	>15	73.8	100.88	73.9	100.86
30-500	<0	0.0%	--	0.0%	--
	0-5	20.5	$3.27	34.3	$2.65
	5-15	79.1	6.43	79.6	6.41
	>15	79.5	7.99	79.5	7.99
500+	<0	0.0%	--	0.0%	--
	0-5	50.0	$6.29	50.0	$6.29
	5-15	79.1	1.67	79.6	1.67
	>15	95.1	2.80	95.1	2.80

Assuming 10 years of post-closure, the present value of clean-up costs for new units ranges from $2.1 million to $18.3 million, with a median of $4.60 million. If post-closure lasts 40 years, the present value of clean-up costs covers a similar range, and the median is slightly lower at $4.59 million.

Corrective action at existing units is modeled in the same way as it is for the other four options. The detail is shown on Table VII-5.

Table VII-28 summarizes the costs of the statutory minimum across size categories given different assumptions about the post-closure care period. The additional years of post-closure care increase the total annualized cost from $702 million to $755 million, an 8 percent increase. The median cost per landfill ranges from $30,700 to $43,300.

Table VII-28

Alternative 4: Annualized Combined Costs by Size; Including Corrective Action

10- and 40-Year Post-Closure Care Periods

(dollars in millions)

		10-Year Post-Closure		40-Year Post-Closure	
Size (TPD)	**Percentage of MSWLFs**	**Annualized Cost**	**Percentage of Total Cost**	**Annualized Cost**	**Percentage of Total Cost**
10	51.3%	$83.6	11.9%	$94.8	12.5%
25	17.0	112.4	16.0	128.0	16.9
75	13.1	92.2	13.1	103.1	13.7
175	7.3	107.4	15.3	113.1	15.0
375	5.5	113.7	16.2	119.7	15.9
750	3.1	77.8	11.1	81.1	10.7
1,500	2.6	115.2	16.4	115.3	15.3
	100.0%[a]	$702.3	100.0%	$755.1	100.0%

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[a] Does not add because of rounding.

Incremental costs per ton are shown on Table VII-29. While these median costs are the lowest among the five regulatory options (the medians range between $7.89 and $9.36 per ton), this option also imposes very high incremental costs (the maximum is $113.19) because of corrective action; without corrective action the maximum cost per ton is only $16.40.

Table VII-29

Alternative 4: Distribution of Incremental Cost per Ton; Including Corrective Action

Incremental Cost per Ton	10-Year Post-Closure	40-Year Post-Closure
$0 - $1	6.8%	5.4%
$1 - $10	59.2	49.2
$10 - $25	19.2	29.6
$25 - $50	1.9	1.7
$50 - $100	10.2	11.4
> $100	2.7	2.7
	100.0%	100.0%

Cost Summary

The combined annualized costs for the five options range from $702 million for Alternative 4 up to $2,844 million for Alternative 2 (Subtitle C design standards). The final rule falls near the lower end of this range, given a range of post-closure care periods. Table VII-30 summarizes the national compliance costs for all options, both with and without corrective action and under our two bounding assumptions on post-closure care. Later in this chapter we will present the results of our adjustment to the lower costs. The adjustment takes into account shifts of municipal solid waste to larger, regional landfills and projected increases in recycling, source reduction, and waste combustion. The relative costs across options are affected by the stringency of the requirements only, since the size distribution across units and the range of remaining lives are constant across options. Corrective action is triggered for some existing units under all options, but is triggered for new units only under Alternative 4.

Table VII-30

Summary of Annualized Compliance Costs for All Options
With and without Corrective Action

(dollars in millions)

10-Year Post-Closure Care Period[1]

	No Corrective Action	Including Corrective Action	Percentage of Total Cost from Corrective Action
Final Rule	$926.2	$975.0	5.0%
Alternative 1 (Proposal)	892.9	941.7	5.2
Alternative 2 (Subtitle C Design)	2,702.4	2,751.2	1.8
Alternative 3 (Categorical Approach)	1,120.8	1,169.5	4.2
Alternative 4 (Statutory Minimum)	177.7	702.3	74.7

40-Year Post-Closure Care Period

	No Corrective Action	Including Corrective Action	Percentage of Total Cost from Corrective Action
Final Rule	$988.8	$1,037.6	4.7%
Alternative 1 (Proposal)	917.9	966.6	5.0
Alternative 2 (Subtitle C Design)	2,794.8	2,843.5	1.7
Alternative 3 (Categorical Approach)	1,183.4	1,232.1	4.0
Alternative 4 (Statutory Minimum)	230.4	755.1	69.5

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[1] Post-closure care is set at 10 years except for the proposal, where it is 30 years.

The ordering of options from least expensive to most expensive is the same, regardless of the post-closure assumptions. The final rule falls in the middle of the five options; the proposal is slightly less expensive (3 to 7 percent) than the final rule, but the categorical and Subtitle C options are much more expensive.

Corrective action is most significant for the statutory minimum because that option imposes no containment or cover requirements so new units trigger corrective action and thus much higher corrective action costs.

Costs per Landfill

We also disaggregated costs into annualized combined costs per landfill to compare the statistics across options. Table VII-31 presents the median, maximum, and minimum costs per unit for the upper and lower estimates. The Subtitle C alternative has the highest incremental median cost (between $163,000 and $177,000 per year), by a factor of more than two over the next highest, the proposal. The final rule is less expensive than the proposal for the median landfills, even though the total cost of the final rule is higher. If we examine the maxima, however, the final rule is the second most expensive option (after Subtitle C). The costs of installing uniform designs at some new facilities account for these high values.

Table VII-31

Annualized Combined Costs per Landfill for All Options

Costs Include Corrective Action

10-Year Post-Closure Care Period[1]

	Minimum Cost per Landfill	Median Cost per Landfill	Maximum Cost per Landfill
Final Rule	$1,700	$44,400	$7,168,900
Alternative 1 (Proposal)	2,900	67,500	6,218,400
Alternative 2 (Subtitle C Design)	3,400	162,900	18,274,500
Alternative 3 (Categorical Approach)	1,600	59,200	5,908,900
Alternative 4 (Statutory Minimum)	1,100	30,700	2,069,300

40-Year Post-Closure Care Period

	Minimum Cost per Landfill	Median Cost per Landfill	Maximum Cost per Landfill
Final Rule	$1,900	$53,700	$7,298,200
Alternative 1 (Proposal)	3,000	71,600	6,266,900
Alternative 2 (Subtitle C Design)	4,500	177,100	18,562,000
Alternative 3 (Categorical Approach)	1,900	68,500	6,038,200
Alternative 4 (Statutory Minimum)	1,300	43,300	2,144,800

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[1] Post-closure care is set at 10 years except for the proposal, where it is 30 years.

Range of Costs per Ton

In Table VII-32, we present statistics on the incremental cost per ton for all of the options under two post-closure care assumptions. The ordering of the options from lowest to highest cost is the same regardless of the length of the post-closure care period. The final rule has the second lowest median cost, approximately 50 percent higher than the statutory minimum, but well below the proposal, the categorical option, and the Subtitle C option. Using maximum costs per ton, the final rule falls in the middle of the options, slightly more expensive than the categorical and proposal options but much less expensive than the statutory minimum or Subtitle C. The incidence of high design costs under the subtitle C option accounts for its high upper bound. Corrective action has a similar effect on the statutory minimum since corrective action is triggered frequently under that option.

Table VII-32

Combined Incremental Cost per Ton for All Options

Costs Include Corrective Action

10-Year Post-Closure Care Period[1]

	Minimum Cost per Ton	Median Cost per Ton	Maximum Cost per Ton
Final Rule	$0.58	$11.61	$48.70
Alternative 1 (Proposal)	0.82	16.74	47.93
Alternative 2 (Subtitle C Design)	0.68	39.64	85.64
Alternative 3 (Categorical Approach)	0.58	14.92	43.94
Alternative 4 (Statutory Minimum)	0.15	7.89	110.83

40-Year Post-Closure Care Period

	Minimum Cost per Ton	Median Cost per Ton	Maximum Cost per Ton
Final Rule	$0.67	$13.82	$52.81
Alternative 1 (Proposal)	0.85	17.82	49.43
Alternative 2 (Subtitle C Design)	0.79	41.50	92.39
Alternative 3 (Categorical Approach)	0.67	17.03	48.05
Alternative 4 (Statutory Minimum)	0.23	9.36	113.19

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[1] Post-closure care is set at 10 years except for the proposal, where it is set at 30 years.

Adjustments to the Costs for Regionalization and for Changes in Future Waste Discards

In Chapter IV, we presented an approach for adjusting the cost of compliance with the regulations downward to take into account underlying shifts in the current waste management practices. Municipal solid waste is gradually moving from more numerous, small landfills into larger, regional units. Concurrently, more source reduction, recycling, and combustion is occurring. Since both of these trends will reduce the costs of compliance with the Subtitle D criteria revisions, we have incorporated these trends into our analysis. Because of the uncertainty associated with modeling the effect of these shifts, however, we have chosen to apply the adjustments to the lower cost estimates presented in this chapter (i.e., the compliance costs assuming shorter post-closure care periods of 30 years for the proposal and 10 years for the other four options including the final rule). The higher cost estimates will remain unchanged, so the actual costs of compliance with the various options will fall somewhere in the range defined by the adjusted lower costs (presented below) and the higher estimates.

Results of Regionalization Analysis

The total amount of waste shifted to larger landfills is reported in Chapter IV. After 20 years of shifting waste from closing units into new, larger ones, 166,400 tons, or 21 percent of the 800,937 tons per day originally landfilled, will shift to larger units. This cumulative total results from the closure of many of the landfills smaller than 125 TPD; the relatively low share of waste shifted results from the correspondingly low share of total MSW handled in these more numerous but small landfills.

The national compliance costs for the 10-year post-closure care scenario in Table VII-30 assume that all of the landfills that close are replaced with units of the same size. The estimates in Table VII-33 show the previous estimates of cost (for the 10-year post-closure scenario) by size category, the cost savings resulting from regionalization 20 years after the effective date, and the adjusted cost estimates. We estimated the savings by calculating the compliance costs before and after the shift for only those landfills that would regionalize. The cost for 10-TPD landfills falls by nearly 75 percent as a result of regionalization; the savings for larger landfills are less significant until we reach the 375-TPD landfills where no shifts are projected to occur. Overall, the shift to regionalization will reduce compliance costs by 22 percent 20 years after the effective date, a savings of $219 million per year.

Table VII-33

Final Rule: Annualized Compliance Costs Adjusted for Regionalization

Annualized Costs 20 Years After Effective Date
(dollars in millions)

Size (TPD)	Annualized Costs[1] Without Regionalization	Savings 20 Years After Effective Date	Adjusted Annualized Costs[1]
10	$62.0	$39.1	$22.9
25	107.7	60.8	46.9
75	160.8	74.2	86.6
175	149.2	44.8	104.4
375	146.9	0.0	146.9
750	154.0	0.0	154.0
1,500	194.4	0.0	194.4
Total	$975.0	$218.9	$756.1

[1] Post-closure care is set at 10 years.

The savings build gradually between now and 20 years after the effective date. As we discussed in Chapter II, many small landfills are projected to close by the effective date; the savings from shifts as of the effective date are $98.4 million, or 45 percent of the $219 million savings achieved by 2011. Five years after the effective date, the cumulative savings rise to $132 million, and ten years after the effective date the savings are $202 million.

Based on the shifts projected for the final rule, we also adjusted the costs for the other regulatory alternatives. Twenty years after the effective date of the regulation, the annual compliance costs for all of the regulatory options will decline by between $158 million and $618 million, as shown in Table VII-34. The final column of the table lists the adjusted annual compliance costs faced by landfill owners in 2011. We have one further adjustment to make, however, which is based on projected decreases in future municipal waste discards.

Table VII-34

Annualized Compliance Costs Adjusted for Regionalization for All Options

Annualized Costs 20 Years After Effective Date
(dollars in millions)

	Annualized Costs[1] Without Regionalization	Savings 20 Years After Effective Date	Adjusted Annualized Costs[1]
Final Rule	$975.0	$218.9	$756.1
Alternative 1 (Proposal)	941.7	211.4	730.3
Alternative 2 (Subtitle C Design)	2,751.2	617.7	2,133.5
Alternative 3 (Categorical Approach)	1,169.5	262.6	906.9
Alternative 4 (Statutory Minimum)	702.3	157.7	544.6

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[1] Post-closure care is set at 10 years except for the proposal, where it is 30 years.

Changes in Future MSW Discards

The adjustment for increased source reduction, recycling, and combustion is made after the regionalization adjustment, and is based on projections of future discards prepared for EPA. As we reported in Chapter IV, we expect a cumulative drop in the quantity of waste landfilled of 18 percent (1 percent per year) 20 years after the effective date of the regulation. Table VII-35 shows the savings estimated 20 years after the effective date and the adjusted cost for each of the regulatory options. This last column of numbers is our best estimate of the annual compliance costs resulting from each of the regulatory options in 2011. As with the regionalization estimate, these cost savings increase gradually over time. These estimates are projections based on EPA data on future MSW discards and intended only as a sensitivity analysis to the results of MWC capacity. The estimates could change significantly if actual capacity brought on-line increases or decreases.

Table VII-35

Annualized Compliance Costs Adjusted for Projected MSW Discards for All Options

Annualized Costs 20 Years After Effective Date
(dollars in millions)

	Annualized Costs[1] After Regionalization	Savings from Waste Diversion	Adjusted Annualized Costs[1]
Final Rule	$756.1	$136	$620
Alternative 1 (Proposal)	730.3	131	599
Alternative 2 (Subtitle C Design)	2,133.5	384	1,750
Alternative 3 (Categorical Approach)	906.9	163	744
Alternative 4 (Statutory Minimum)	544.6	98	447

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[1] Post-closure care is set at 10 years except for the proposal, where it is 30 years.

Conclusion

The final results of the cost analysis are shown in Table VII-36. We have incorporated into the lower cost estimates a relatively short post-closure care period of 10 years for all of the options except Alternative 1, which has a 30-year post-closure care period. The lower estimates are also adjusted to take account of shifts in the quantity of waste landfilled due to regionalization and increased reliance on source reduction, recycling, and combustion 20 years after the effective date. The higher cost estimates reflect a 40-year post-closure care period and none of the adjustments for regionalization or future waste diversion.

Table VII-36

Annualized Compliance Costs for All Options

Annual Costs 20 Years After the Effective Date
(dollars in millions)

	10-Year Post-Closure Care Scenario[1]	40-Year Post-Closure Care Scenario[2]
Final Rule	$620	$1,037.6
Alternative 1 (Proposal)	599	966.6
Alternative 2 (Subtitle C Design)	1,750	2,843.5
Alternative 3 (Categorical Approach)	744	1,232.1
Alternative 4 (Statutory Minimum)	447	755.1

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[1] Post-closure care is set at 10 years except for the proposal, where it is 30 years; shifts to larger landfills and to waste combustion are factored into the estimates.

[2] No adjustments are made for shifts to larger landfills or for increased diversion from landfills.

Alternatives 2 and 3 are the most expensive options because they impose the most stringent design requirements on landfills. The final rule and proposal both incorporate performance standards into the decision about which designs are required, but the final rule does not rely exclusively on performance standards; it establishes a uniform design standard that is self-implementing for landfills in unapproved states. Finally, the statutory minimum imposes no design standards on landfills; it achieves protection of human health and the environment through ground-water monitoring and corrective action requirements only.

We project that annual compliance costs for each option in 2011 would fall between these boundaries. The lower cost estimates incorporate optimistic assumptions about the willingness of landfill owners to shift to regional landfills and about diversion of waste from landfills because of increased source reduction, recycling, and combustion. If no additional shifts occur in the future, the costs will fall at the upper end of the range. One final factor that affects these estimates is that other flexibility incorporated in the final rule is not captured. For example, states could allow post-closure care to vary over a wider range and could allow landfills to comply with the performance standard at an alternative point of compliance farther from the landfill boundary. This would decrease the costs associated with new landfill design. Finally, the RIA does not incorporate the exemption for small (remote) landfills in the final rule. While this would not affect aggregate costs significantly, it may change the results of the economic impact analysis and regulatory flexibility analysis in Chapter VIII.

VIII. Results: Economic Impacts

Disposal of solid waste is one of many functions performed by local governments. Many municipalities own and operate their own landfills, some participate in regional landfills operated by counties or nearby communities, others contract with private landfills, and some rely on other means to dispose of MSW such as participating in resource recovery facilities. All communities incur some cost for solid waste disposal regardless of whether this cost appears as a line item in a municipality's budget or as a fee or tax that its citizens pay back to the municipality, to a private hauler, or to another government.

This chapter focuses on the impacts to communities and households that are served by MSWLFs using the methodology described in Chapter V and the costs reported in Chapter VII. The chapter begins with a discussion of the impacts of the regulatory options, focusing first on the final rule and then on a comparison of the alternatives. Next we discuss how the impacts vary given the baseline financial capability of municipalities. The final part of the chapter presents an analysis of impacts on small entities.

As introduced in Chapter V, the impact analysis focuses on economic impacts on municipalities themselves and impacts on the households served by these municipalities. To measure impacts on municipalities, we compared the incremental costs of the options with each municipality's total current expenditures to calculate cost as a percentage of expenditures (CPE). We measured impacts on households by calculating incremental cost per household (CPH) and cost per household as a percentage of median household income (CPMHI). In all cases, we present the impacts of the combined costs on both new and existing landfills, including corrective action costs. Communities are classified as having severe impacts if CPE or CPMHI exceeds 1 percent or if CPH exceeds $220. Chapter V explains how these threshold levels were selected.

Impact Analysis Results

Impacts of the Final Rule

The discussion below summarizes the impacts of the final rule, using the cost estimates from Chapter VII, which assume a 40-year post-closure care period. (A short discussion of the sensitivity of the impact results to lower-cost scenarios is presented later in the chapter.) These costs will also provide a reasonable upper bound on impacts because they assume 40 years of monitoring and other post-closure care, no shift in waste to regional landfills over time, no increased waste diversion, and no move toward better siting (and therefore less expensive designs under the performance standard) as a result of compliance with the regulations.

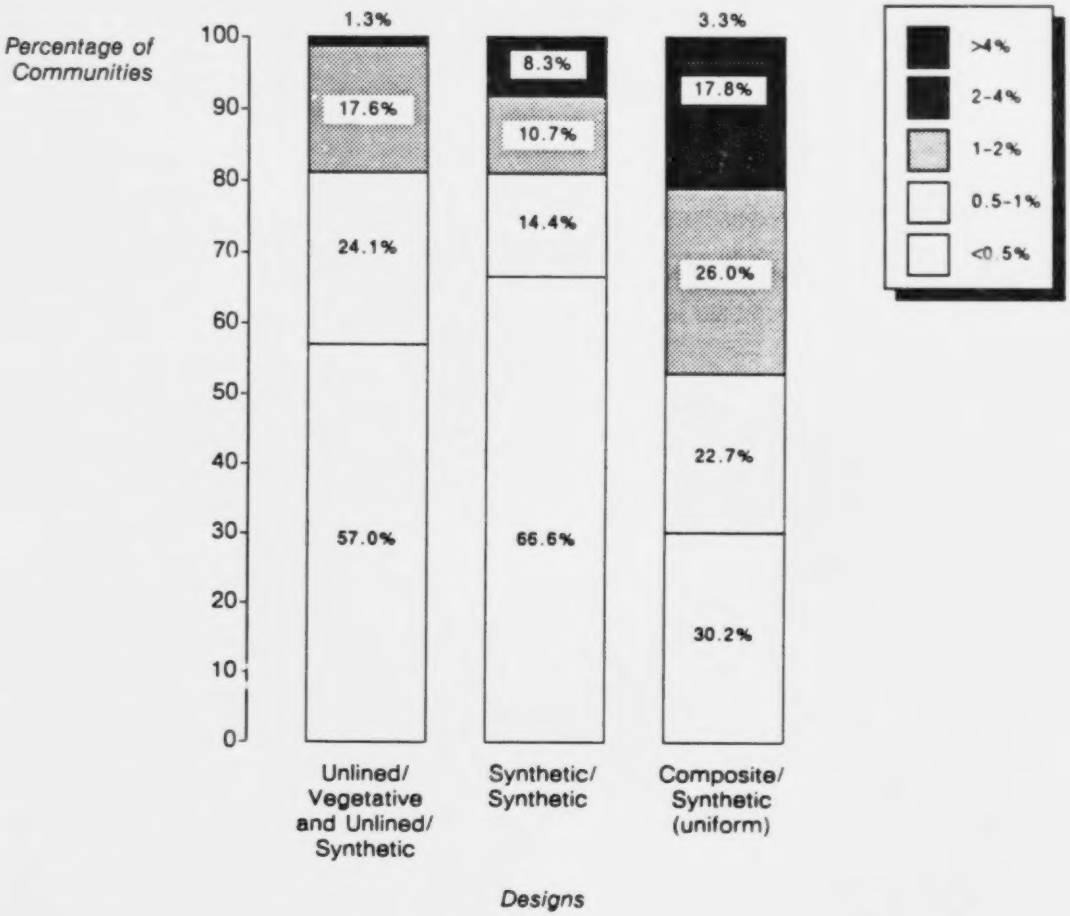
The median cost as a percentage of expenditures under the final rule is 0.49, so the median community will face an increase of less than 0.5 percent of current expenditures for compliance with the rule (i.e., half of the jurisdictions will face higher CPE and half lower). Twenty-seven percent (or 7,822 communities) will face increases of more than 1 percent with a maximum increase of 7.2 percent. Nearly 70 percent of the 7,822 communities (5,456 communities) have increases of 1 to 2 percent. Another 20 percent of these 7,822 communities have CPE between 2 and 3 percent.

Design plays a major role in determining impacts. As shown in Figure VIII-1, over 47 percent of the communities served by landfills assigned to the uniform (composite-lined) design incur significant increases in CPE. Only 19 percent of communities served by unlined landfills after the rule face an increase of greater than 1 percent in CPE. Of those communities facing increases of over 2 percent, 74 percent use landfills with the composite liner design.

Figure VIII-1

Cost as Percentage of Expenditures

By Design for Final Rule (40-Year Post-Closure)

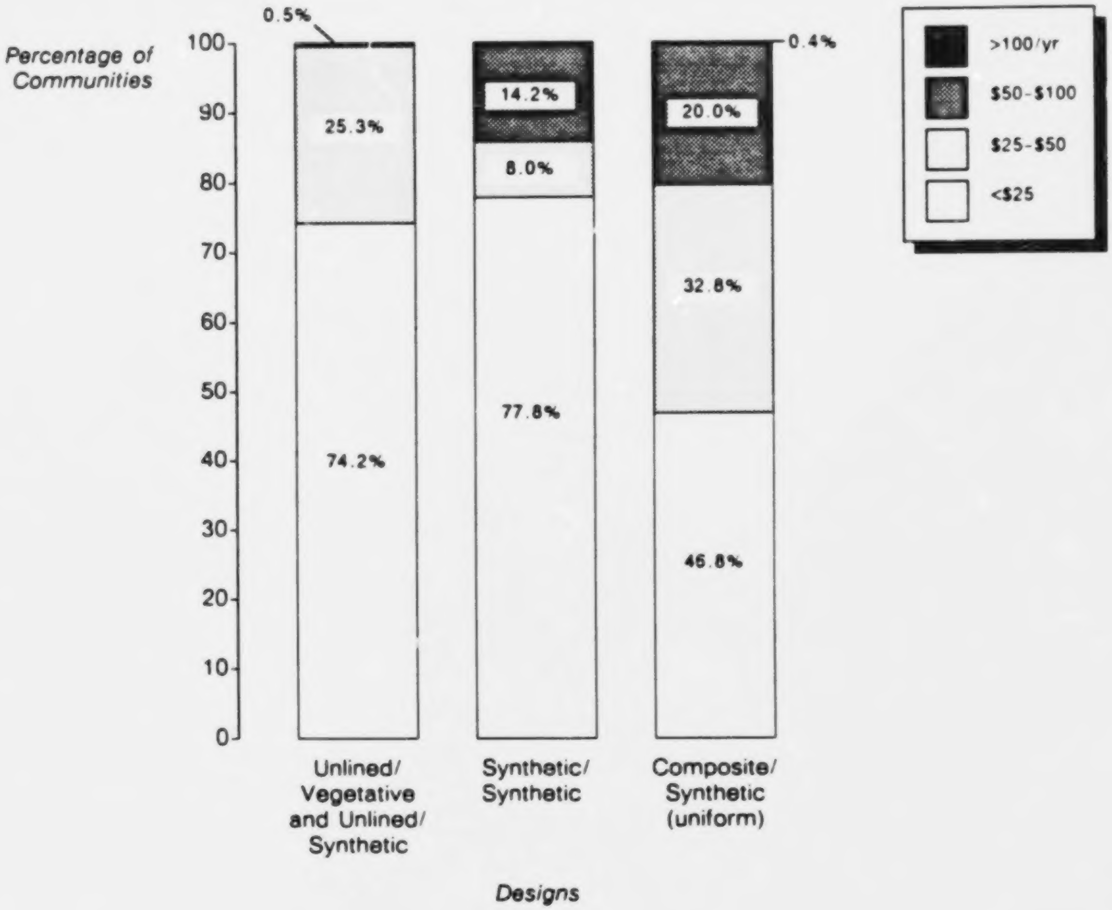


The CPH ranges between $0.82 and $175 per year across the 29,000 communities analyzed. The median community in the analysis faces an incremental cost per household of $15.98. Only 0.1 percent of communities see increases in the moderate range of greater than $100 per household; none exceed the significant impact threshold of $220. As with CPE, design requirements affect the distribution of incremental costs per household. Figure VIII-2 shows that the incidence of CPH greater than $50 increases from 0.5 percent of communities using unlined landfills to 14.2 percent for communities with synthetic-lined landfills to 20.4 percent for communities with composite-lined landfills.

Figure VIII-2

Cost per Household per Year

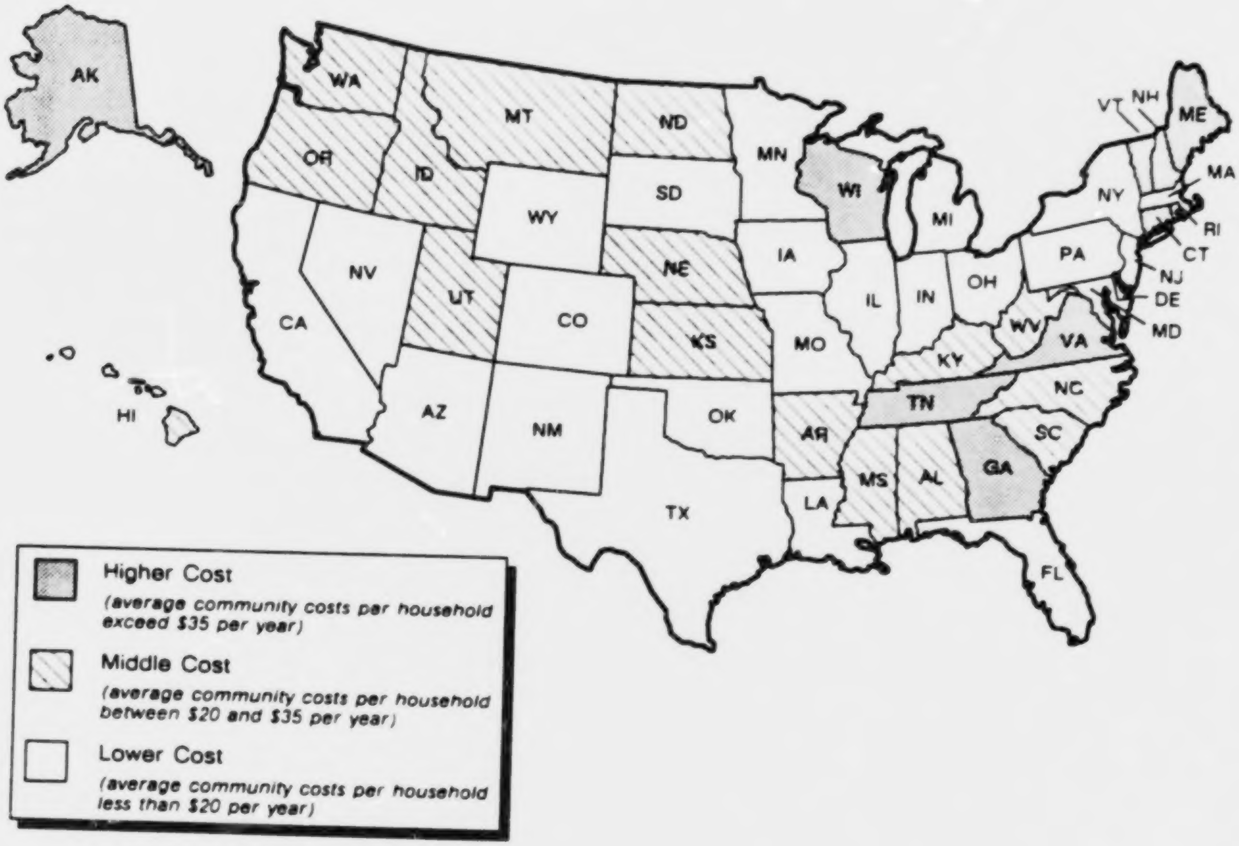
By Design for Final Rule (40-Year Post-Closure)



CPH also varies by state, as depicted in Figure VIII-3. The states with the highest CPH are in New England plus Virginia, Tennessee, Georgia, Wisconsin, and Alaska. All of these states have landfills with average sizes well below the national average, which will increase their compliance costs. In addition, all of these states have annual precipitation well above the national average, so landfills in these states are more likely to require more expensive designs.

Figure VIII-3

Average Community Costs per Household by State for the Final Rule (40-Year Post-Closure Care)


AK
WA
MT
ND
MN
VT
NH
ME
OR
ID
SD
WI
MI
NY
MA
WY
RI
CT
PA
IA
NE
NJ
NV
UT
IL
IN
OH
DE
CA
CO
KS
MO
WV
VA
MD
KY
NC
HI
AZ
NM
OK
AR
TN
SC
GA
MS
AL
LA
TX
FL
Higher Cost
(average community costs per household exceed $35 per year)
Middle Cost
(average community costs per household between $20 and $35 per year)
Lower Cost
(average community costs per household less than $20 per year)

Impacts on households also depend on who owns the landfill that serves those households. Table VIII-1 indicates the number of communities and landfills by major ownership category: county, city, village or town, private, and other. ("Other" covers landfills owned by non-local governments, including special districts, states, and the federal government.) The distribution of communities by owner type looks somewhat different from the same distribution of landfills because county-owned and private landfills tend to serve a larger number of communities than city or town landfills. The table indicates that communities served by village or town landfills have much higher CPH than average. These landfills tend to serve only one or two communities and are commonly very small, thus the higher CPH. Communities served by private landfills, on the other hand, have lower than average CPH. These landfills serve many communities and tend, on average, to be larger than publicly owned landfills.

Table VIII-1

Number of Communities and Landfills by Type of Owner: Final Rule

(40-year post-closure care)

Owner	Communities		Landfills		Average Number of Communities per Landfill	Average Community Cost per Household
	Number	Percentage	Number	Percentage		
County	10,618	37%	1,760	29%	6.0	$23
City	6,622	23	1,743	29	3.8	24
Village or Town	2,115	7	1,182	20	1.8	43
Private	8,556	30	912	15	9.4	13
Other	1,087	4	427	7	2.6	16
Total	28,998[a]	100%[b]	6,024[a]	100%	4.8	$22

[a] Data are missing for 10 landfills in 19 communities.
[b] Totals may not add to 100 percent because of rounding.

If we examine CPMHI, we also find a range of increases, from less than 0.01 percent to 1.26 percent, but the median community has a CPMHI of 0.074 percent. Only 0.07 percent of all communities exceed the significant impact threshold of 1 percent, and only 0.51 percent have a CPMHI above 0.5 percent. The communities exceeding the threshold have populations between 5,000 and 10,000 and are served by 10-TPD landfills that must comply with either the uniform design standards or the synthetic/snythetic design. The impacts of the final rule are concentrated more on communities (as shown by the CPE) than on individual households in those communities. Typical households would see only modest increases in their costs of disposal as a result of the final rule, and households in 93 percent of all communities would see increases of less than 0.25 percent of median household income.

Comparison of Regulatory Alternatives

The discussion of the four regulatory alternatives focuses on the numbers of communities that exceed the threshold levels of each of the three economic impact measures. Impacts are highest under Alternative 2 and lowest under Alternative 4, regardless of the impact measure used. At the same time, Alternative 4 imposes the most severe impacts (i.e., the highest CPE, CPH, and CPMHI) because of the high cost of corrective action. The impacts of the final rule are also briefly summarized below to facilitate comparison with the alternatives.

Table VIII-2 indicates the percentage of communities and people whose CPE will exceed 1 percent under the final rule and each of the four alternatives. Under Alternative 2, 64 percent of communities have CPE exceeding 1 percent, while under Alternative 4, only 14 percent of all communities fall in this category. The percentage of municipalities with costs above 1 percent of current expenditures under the proposal is 29 percent and is similar to the impacts under the final rule. Because most of these affected communities are small, the percentage of total U.S. population that resides in these communities is much smaller than the percentage of communities affected.

Table VIII-2

Municipalities and People with Compliance Costs Exceeding One Percent of Current Expenditures

(40-year post-closure care)

	Municipalities		People	
Regulatory Option	**Number**	**Percentage of Total**	**Number (millions)**	**Percentage of Total**
Final Rule	7,822	27.0%	23.6	10%
Alternative 1 (Proposal)	8,515	29.3	22.3	9
Alternative 2 (Subtitle C)	18,581	64.0	68.5	29
Alternative 3 (Categorical)	10,319	35.6	27.9	12
Alternative 4 (Statutory Minimum)	4,168	14.4	10.1	4

Source: TBS calculations.

Using data from EPA's MSWLF survey, we estimate that the typical community currently spends approximately 0.5 percent of its budget on solid waste disposal. For many communities, the incremental costs of complying with these regulatory options will exceed 1 percent of their current expenditures, a 200 percent increase in their solid waste disposal costs.

Table VIII-3 presents descriptive statistics for the CPE measure under all of the regulatory options. The CPE for the median community varies by a factor of four from the statutory minimum to the Subtitle C option. With the exception of the statutory minimum, all the alternatives have higher median CPE than the final rule. The maximum CPE is as high as 20 percent for the statutory minimum because of corrective action costs, but most communities have much lower CPE.

Table VIII-3

CPE for All Regulatory Options

(40-year post-closure care)

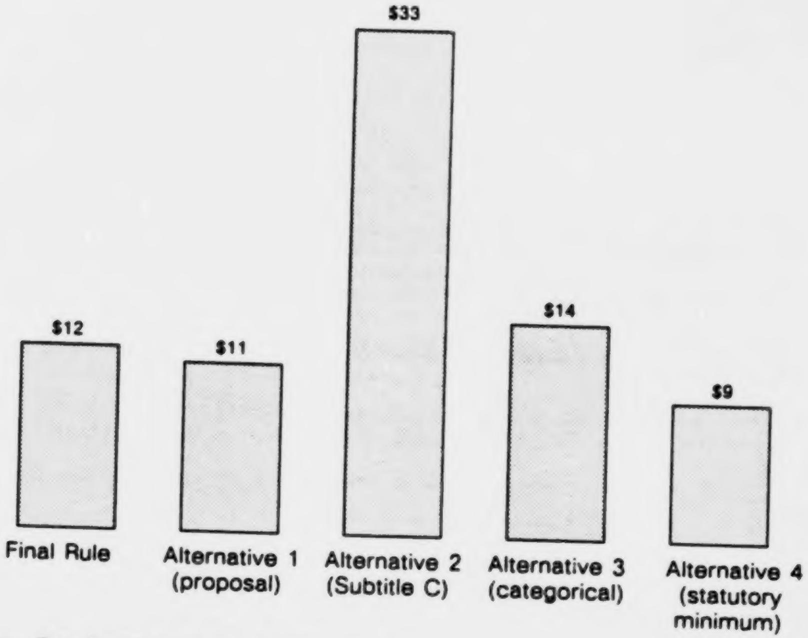
	Minimum	Median	Maximum
Final Rule	0.01%	0.49%	7.17%
Alternative 1 (Proposal)	0.01	0.54	5.80
Alternative 2 (Subtitle C)	0.02	1.48	13.06
Alternative 3 (Categorical)	0.01	0.64	7.19
Alternative 4 (Statutory Minimum)	<0.01	0.35	19.74

Turning to the impacts on households, Figure VIII-4 summarizes the impacts of the regulatory options on households nationwide. Total compliance costs divided by total U.S. households ranges from $9 under Alternative 4 to $33 under Alternative 2. Incremental CPH under the proposal averages $11 across all households compared to $12 for the final rule.

Figure VIII–4

Average Incremental Cost per Household Nationwide

(dollars per year)



Note: Data based on 40-year post-closure care.

Source: TBS calculations.

If we compare CPH among the 29,017 individual communities instead of all households, the median ranges from $12 for Alternative 4 to $46 for Alternative 2. In the majority of communities, current solid waste disposal costs per household range from $10 to $30 per year (excluding collection and transportation costs), with fewer than 10 percent of all communities' CPH exceeding $50 per year, according to the MSWLF survey. Thus, even under the least expensive alternative, CPH will likely increase by 50 percent for many communities. As mentioned before, the reported costs in the survey may be too low because they may not include amortization of capital costs. As a result the percentage increase would be smaller. Table VIII-4 presents statistics on CPH in individual communities. Alternatives 2 and 4 exceed the severe impact threshold of $220 and, under Alternative 4, 1.0 percent of communities exceed that threshold because of corrective action. Because relatively few communities exceed the severe impacts threshold, the table also reports the number and percentage of communities exceeding the moderate impact threshold of $100. This statistic shows a dramatic increase in moderate impacts under Alternative 2.

Table VIII-4

CPH for All Regulatory Options

(40-year post-closure care)

				Communities Exceeding $100 CPH	
	Minimum	**Median**	**Maximum**	**Number**	**Percentage of Total**
Final Rule	$0.82	$15.98	$174.56	33	0.1%
Alternative 1 (Proposal)	1.06	17.24	137.36	34	0.1
Alternative 2 (Subtitle C)	1.21	45.56	305.39	5,841	20.1
Alternative 3 (Categorical)	0.82	21.33	158.84	33	0.1
Alternative 4 (Statutory Minimum)	0.29	12.43	361.57	1,701	5.9

Cost per household as a percentage of MHI exceeds 1 percent under all the options except the proposal (Table VIII-5). EPA's construction grants program uses a threshold of 1 percent for this measure to identify communities that might experience difficulties affording new wastewater treatment facilities. In our analysis, fewer than 0.1 percent of communities exceed the threshold under the final rule and Alternative 3. Seven-tenths of one percent exceed the threshold under Alternative 2 and 1.5 percent exceed the threshold under Alternative 4. The maximum CPMHI for any community was 2.2 percent under Alternative 2 and 2.6 percent under the statutory minimum (because of corrective action).

Table VIII-5

CPMHI Across All Regulatory Options

(40-year post-closure care)

				Communities Exceeding One Percent CPMHI	
	Minimum	**Median**	**Maximum**	**Number**	**Percentage of Total**
Final Rule	<0.01%	0.07%	1.26%	20	<0.1%
Alternative 1 (Proposal)	0.01	0.08	1.00	0	0.0
Alternative 2 (Subtitle C)	0.01	0.21	2.21	189	0.7
Alternative 3 (Categorical)	<0.01	0.09	1.15	20	<0.1
Alternative 4 (Statutory Minimum)	<0.01	0.05	2.62	426	1.5%

Comparison of Impacts and Financial Capability

Baseline Financial Capability Results

The first step in the municipal financial capability analysis is to assess the current financial health of local governments by calculating a composite financial rating or score. This score combines information on communities' current revenue, expenditures, debt, and demographic characteristics into a single point estimate of financial capability. Based on this measure, we classified communities as financially strong, average, or weak.[1] Twenty-seven percent of all communities were classified as strong, 49 percent as average, and 24 percent as weak.

Weak governments, which typically have relatively high levels of debt and expenditures, low income, and a high fraction of the population below the poverty level might have difficulty coping with even moderate increases in expenditures for solid waste disposal. Strong communities, on the other hand, typically have the economic base and flexibility in their budgets to accommodate all but the most severe levels of economic impact.

The baseline results are calculated on all 29,017 government jurisdictions contained in the database described in Chapter V. The actual analysis of impacts was conducted on the subset of 1,810 of these communities that are served by the landfills that responded to the MSWLF facility survey. Comparison of financial capability scores for the subset and the entire database indicates that the sample of communities included in the analysis is representative of both the size and financial health of the entire population of local governments. As Chapter V describes, the results of the economic impact analysis on this subset of communities were also weighted based on community size and financial capability to compensate for any minor differences between the characteristics of the sample and the characteristics of all local governments.

If we look at the total *number* of governments that are relatively weak (6,973), 75 percent of them are small (less than 5,000 people), so the majority of weak governments are also small. This pattern, however, reflects the same distribution in the entire sample, that is, 77 percent of all 29,017 governments are small.

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[1] The score can take on any value from 1 through 5. Communities with scores of less than 2.6 were characterized as weak; those with scores greater than 3.2 as strong; and the remainder as average.

Figure VIII-5 summarizes the relative financial capability ratings for all governments by the population they serve. It displays the *percentage* of governments that are classified as strong, average, or weak in each of the population size categories. The chart shows that larger governments are more likely to be classified as weak than smaller ones. Small communities' ratings differ, however, depending on whether or not they are in rural areas. We will address this below.

Figure VIII-5

Financial Capability Rating for Governments

By Size Categories

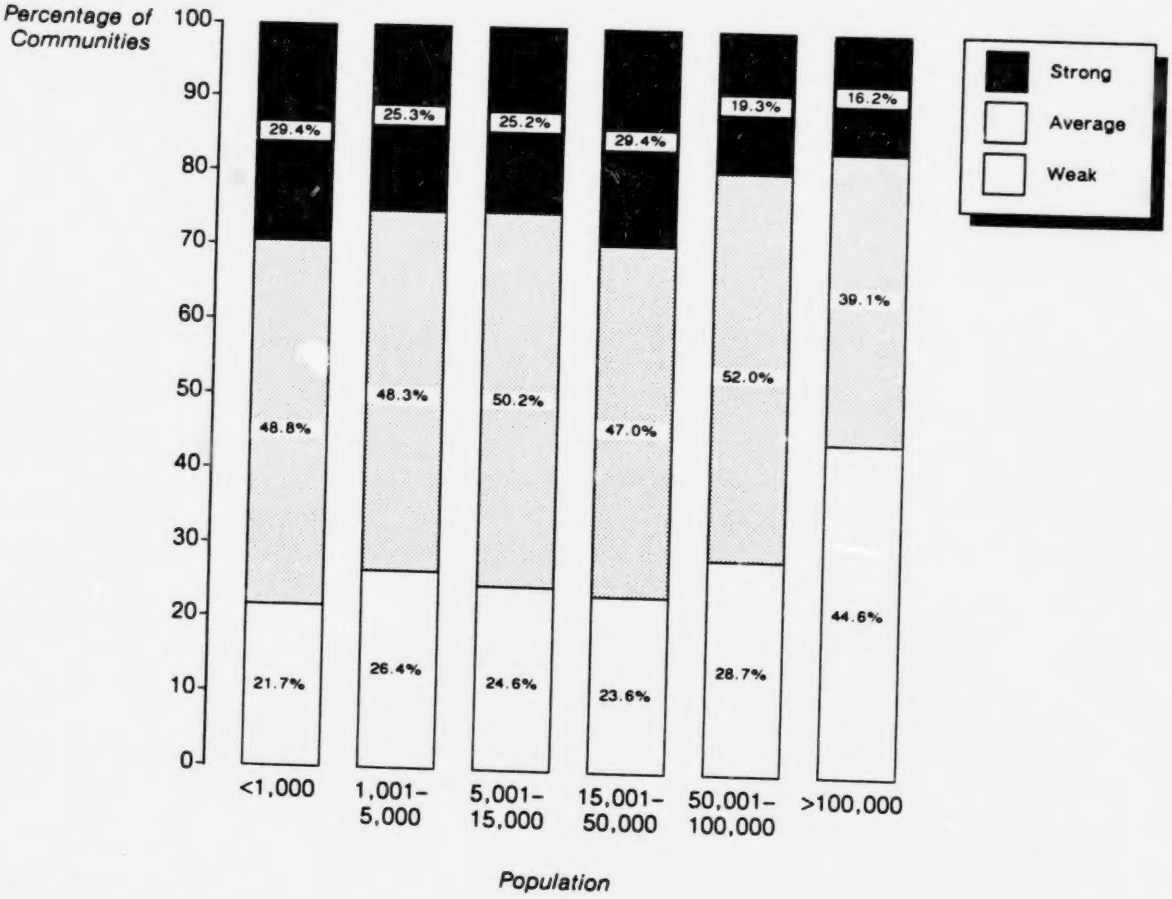
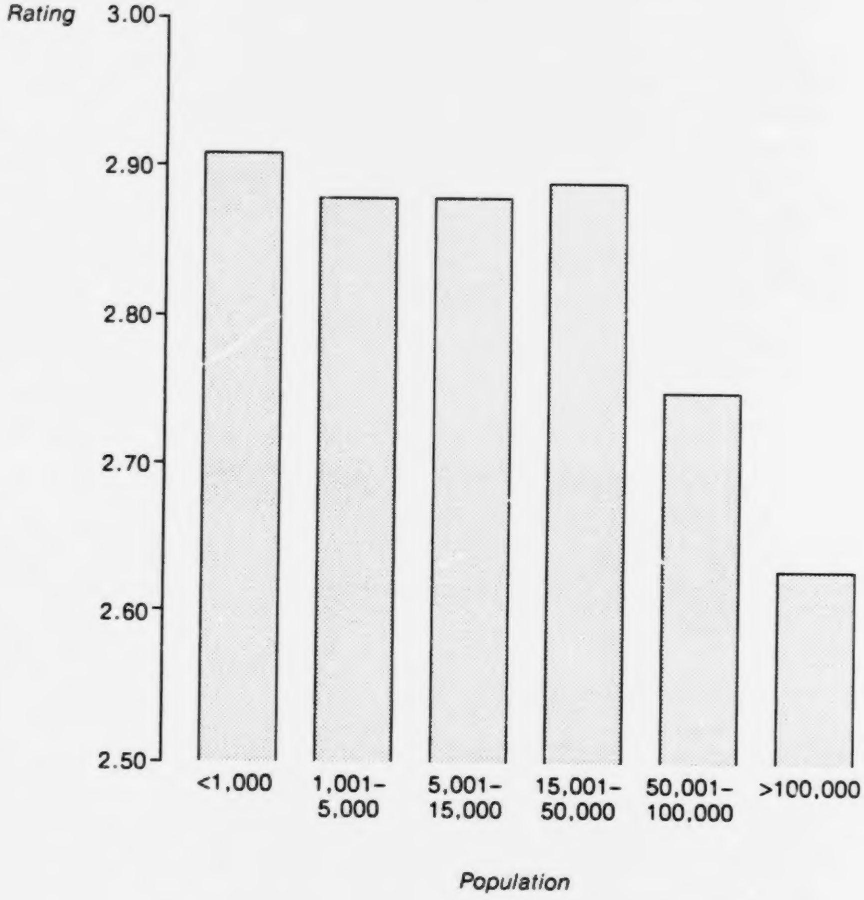


Figure VIII-6, which summarizes the average financial capability rating for each size category, shows the same pattern. Communities with fewer than 1,000 people have the highest scores, and those serving more than 100,000 people the lowest.

Figure VIII-6

Average Financial Capability Rating for Governments

By Size Categories



For the largest communities (primarily cities over 100,000 people), the financial scores are often quite low because their governments provide services to large numbers of people who work or live in the community. Thus, these governments usually spend more and incur more debt per person than smaller jurisdictions. The pattern of small communities' spending is different. For example, they issue a much lower percentage of debt to finance utility expenditures such as water, electric, or transit services than do larger communities. Larger cities are also more likely to have low MHI, and many are losing population to neighboring suburbs; both factors would tend to lower their financial scores.

The financial capability scores across community sizes tend to fall in a fairly narrow range around the average score of 2.89, with few exceptions. Within these community sizes, however, the financial capability scores differ between rural and urban communities. Small communities especially are split between those that are rural and those that are suburban (i.e., associated with a neighboring urban area). We expect that the financial capability scores of rural small towns will be lower than for suburban communities. The rural/urban split was defined by population densities with 40 people per square mile as the cutoff (midway between the national average of 64 and the average of "rural America" of 17 people per square mile defined by the *Statistical Abstract*[2]). Ninety-two percent of rural communities have populations less than 5,000.

Figure VIII-7 shows that for the smallest communities, rural towns are much weaker than suburban towns with a rating of 2.8 versus 3.1. The difference is less pronounced for communities of between 1,000 and 5,000. From these results we expect small rural communities to be adversely affected by the regulations more often than small suburban communities.

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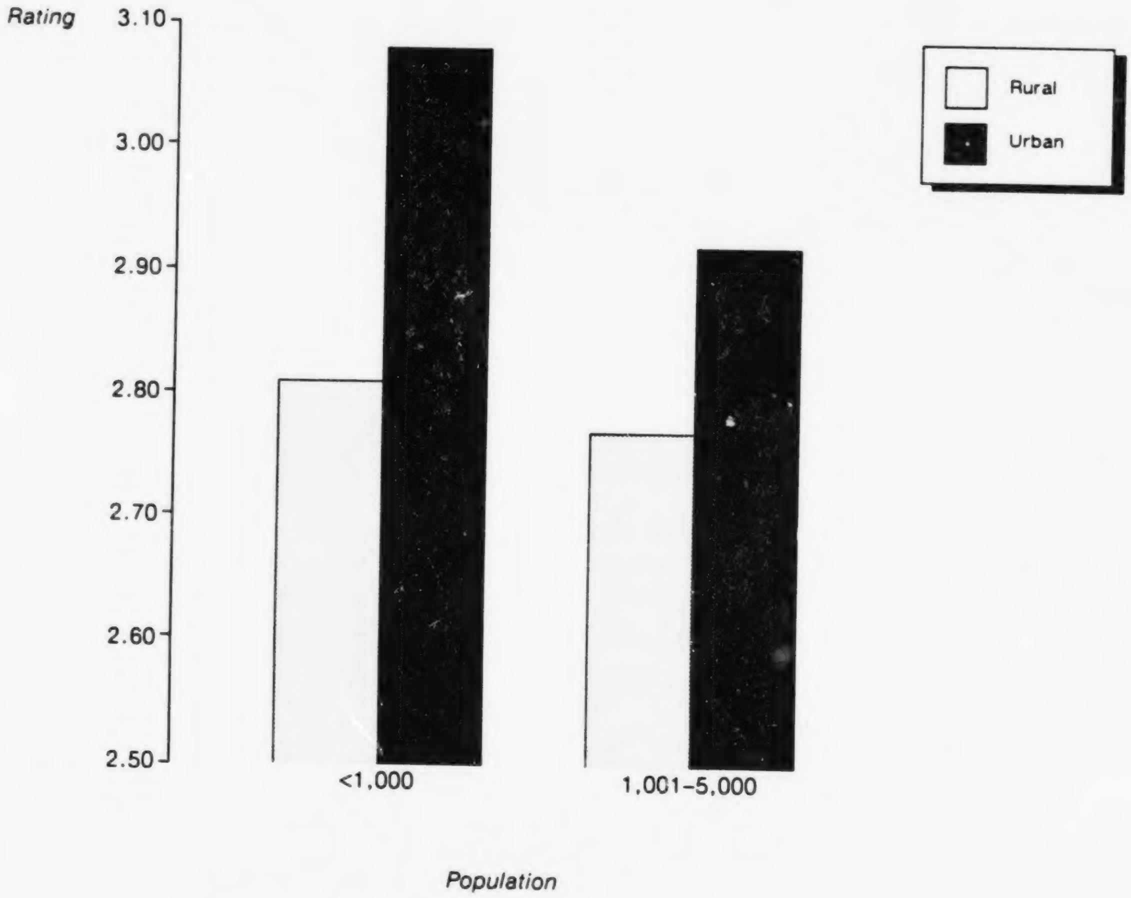
[2] *Statistical Abstract of the United States 1987*, U.S. Department of Commerce, Bureau of the Census.

Figure VIII-7

Financial Capability Rating for Rural Versus Urban Areas

(All Communities Smaller Than 5,000 People)



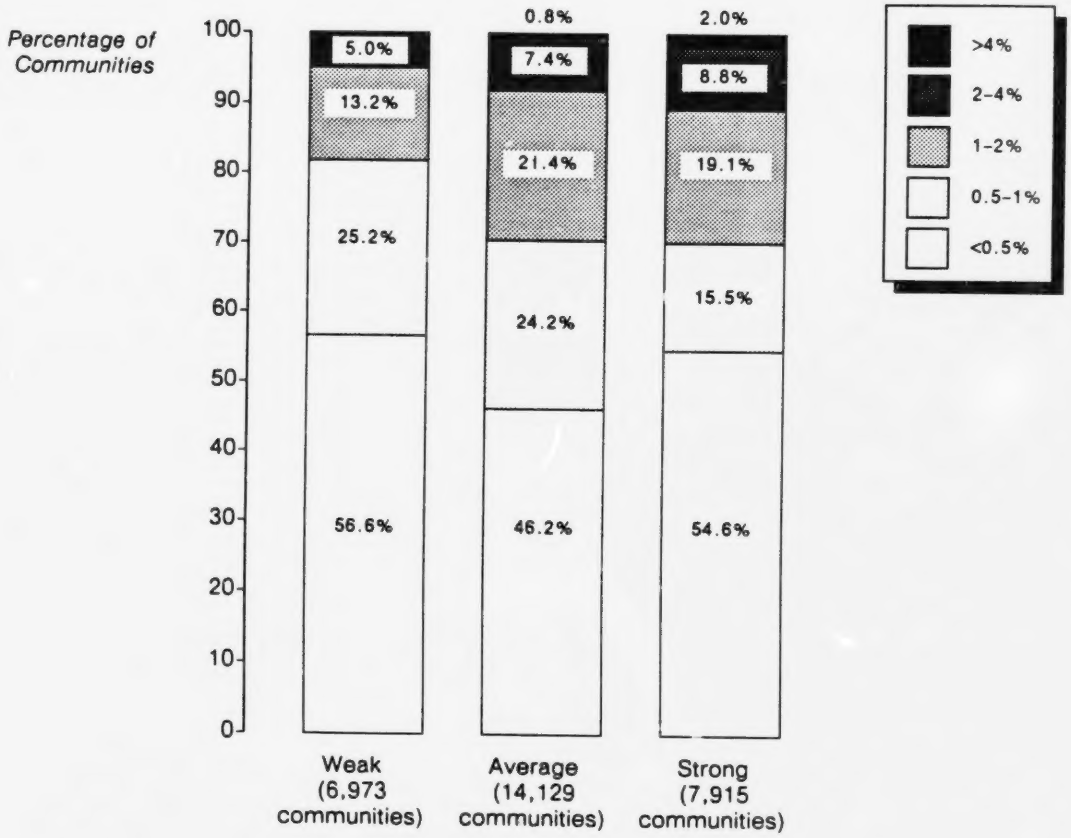
Source: TBS calculations.

Impacts and Financial Capability

The financial capability baseline discussed above enables us to investigate whether communities that incur relatively high costs as a result of the final rule are weak or strong financially. Figure VIII-8 presents the distribution of CPE by communities' financial capability score. The figures show that weak communities are somewhat less likely to face large increases in expenditures than average and strong communities. CPE exceeds 1 percent in 30 percent of all strong and average communities and in 18 percent of all weak communities.

Figure VIII-8

Incremental Cost as Percentage of Expenditures
for Final Rule (40-Year Post-Closure Care)
By Financial Capability Category



Source: TBS calculations.

The major factor accounting for these results is that a large community is more likely to be categorized as weak than a smaller community (even though in an absolute sense, small weak communities outnumber large weak communities). This is especially true for communities in urban areas where the smallest towns tend to have strong scores. The large communities in these areas are served by large landfills, because regulatory costs exhibit substantial returns to scale, these large (weak) towns are less likely to experience severe impacts.

The majority of the weak communities with significant increases in CPE are very small communities: 62 percent (782 of the 1,271 affected communities) have fewer than 1,000 people. Eighty-nine percent (1,132 communities) have populations below 5,000 people. (The relationship between community size and economic impacts is discussed in more detail in the Regulatory Flexibility section of this chapter.)

Of these 1,271 communities, 55 percent are in rural areas, indicating that the rural/urban split is also an important issue in the relationship between financial strength and impacts. For example, Figure VIII-8 shows that the final rule resulted in severe impacts for 18 percent of weak communities. For rural communities only, the fraction of weak communities with severe impacts is 27.9 percent, and the corresponding fraction for urban communities is only 9.9 percent. As we expected, rural communities dominate those with the highest impacts and weak financial capability.

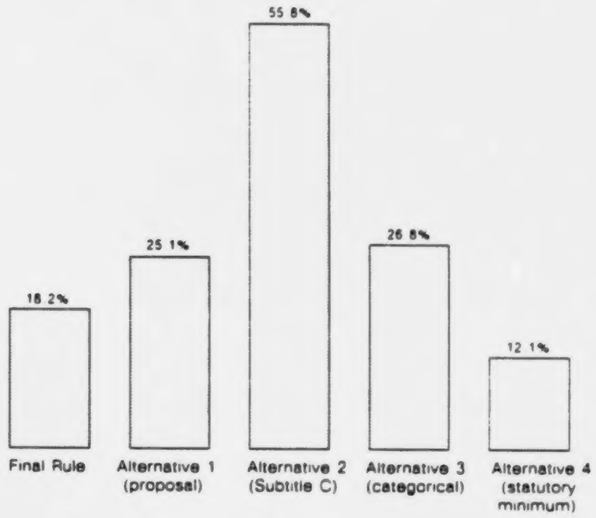
The two cost per household measures (CPH and CPMHI) indicate that the final rule will not have a serious impact on the households served by these communities. Only 20 communities (0.07 percent of the total) face CPMHI above 1 percent. The severe impacts threshold for CPH is not exceeded under the final rule. One-tenth of one percent of communities exceed the $100 moderate impact threshold under the final rule.

Comparison of Impacts and Financial Capability Across Options

The results of the economic impact analysis indicate that most of the options impose significant impacts on at least some communities' expenditures for MSW disposal. Among communities with weak financial capability, 1,271 communities (18.2 percent) face significant increases in CPE from the final rule.[3] Figure VIII-9 shows that all of the regulatory alternatives except the statutory minimum impose significant impacts on a larger share of weak communities than the final rule does. Across all options, between 12 percent and 56 percent of the weak communities will exceed the CPE threshold of 1 percent. Table VIII-2, however, indicates that although a large number of communities fall into the severe impact category for CPE, a far smaller percentage of the population is affected because these communities tend to be the smaller ones.

Figure VIII-9

Percentage of Weak Communities with Compliance Costs
Greater Than 1 Percent of Expenditures
All Options: 40-Year Post-Closure Care



Source: TBS calculations.

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[3] This discussion of impacts on weak communities focuses primarily on CPE because the incidence of CPH and CPMHI among weak communities is negligible across options. Only Alternatives 2 and 4 exceed the CPH threshold for 0.3 and 1.7 percent of weak communities, respectively. The final rule and Alternative 3 impose CPMHI above 1 percent on 0.3 percent of weak communities; Alternative 2 causes 0.9 percent of the weak communities to exceed the threshold; and Alternative 4 drives 2.6 percent of weak communities over the threshold.

Several factors will tend to mitigate the actual impact of the options on communities with high CPE. One important factor is the relatively small proportion of the municipal budget that is usually devoted to MSW disposal. Although CPE greater than 1 percent indicates that MSW disposal expenditures may double in many communities, even after regulation these expenditures will represent less than 2 percent of the total municipal budget in most communities. For comparison, the average community in the United States currently spends 36 percent of its budget on education, 5 percent for police protection, 3 percent for sewage disposal, 2 percent for fire protection, and 1 percent for sanitation services other than sewage, including solid waste collection and disposal and street cleaning (see Table V-1). Although it may be difficult for communities to cope with large percentage increases in MSW disposal costs in the short run, once the initial adjustment is made, these costs should be easier for communities to absorb because they comprise a very small portion of communities' total budgets.

It is also important to recognize that the costs used in the economic impact analysis represent reasonable upper-bound estimates. Several opportunities for communities to reduce regulatory costs are presented in Chapter VII; together, they will reduce annual compliance costs for the final rule to $620 million after 20 years, 40 percent below the higher estimate of $1,038 million. These opportunities include participation in larger, regional landfills that have lower incremental regulatory costs and shifts to source reduction, recycling, and combustion. In addition, these costs assume 40 years of post-closure care and do not incorporate the better siting of landfills that the performance standard will encourage.

Because of the approach we used to adjust the costs, we are unable to provide a complete analysis of economic impacts assuming the lower-bound compliance cost estimates after regionalization and combustion are considered. We did, however, analyze how regionalization would reduce impacts for those communities hardest hit by the final rule. As described above, 7,822 communities incur CPE above 1 percent. Since the regionalization analysis was conducted by size categories of landfills, we assessed the landfill sizes in these 7,822 communities. Approximately 60 percent of the landfills were in the 10-TPD category and none were in the 750- or 1,500-TPD categories.

Based on waste shifts summarized in Table IV-4, most of the waste landfilled in these smaller units will shift to larger units after 20 years. Once the waste is shifted to these larger landfills, the compliance costs will decline and we expect that the impacts will no longer be significant. We assumed that if 50 percent of the waste shifts out of a certain size category, the same fraction of communities will face lower economic impacts.

Table VIII-6 reports the percentage of waste in each of the smaller size categories that will shift to larger landfills (375-TPD and larger) 20 years after the effective date of the regulation. The table also shows the number of communities with significant impacts before and after the regionalization shifts. This analysis suggests that the number of communities that face significant impacts from the final rule will drop by nearly 84 percent after 20 years, so the estimates presented in this chapter represent reasonable upper bounds on the impacts that communities will face. Given the estimates in Table VIII-6, only about 4 percent of communities would face significant impacts after 20 years instead of 27 percent.

Table VIII-6

Effect of Regionalization on Communities with Significant Impacts[1]

Landfill Size (TPD)	Percentage of Waste Shifted to Larger Landfills After 20 Years	Communities With Significant Impacts: Effective Date: Number	Effective Date: Percentage of Total[2]	After 20 Years: Number	After 20 Years: Percentage of Total[2]
10	85%	4,689	16.2%	703	2.4%
25	83	1,397	4.8	237	0.8
75	88	994	3.4	119	0.4
175	80	645	2.2	129	0.4
375	0	97	0.3	97	0.3
	21%	7,822	27.0%	1,285	4.4%

---

[1] Final rule CPE exceeds 1 percent.

[2] Calculated as a percentage of all 29,017 primary jurisdictions; totals may not add because of rounding.

Other mitigating factors exist as well. Some of the communities are in states with existing regulatory requirements that we did not incorporate into our analysis. These communities may face lower compliance costs because they already meet certain provisions of the revised landfill criteria. Finally, the cost model assumptions imply that communities set aside money during the operating life of the landfill to provide for post-closure care (i.e., post-closure compliance costs are paid from revenues generated during the operating life). Although private operators are likely to do this, publicly-owned landfills may have opportunities (depending on how financial responsibility requirements are met) to reduce impacts by deferring payment for post-closure care until the costs are actually incurred.

Regulatory Flexibility Analysis

Introduction

The Regulatory Flexibility Act requires federal regulatory agencies to determine whether regulations will have a significant economic impact on a substantial number of small entities. When such impacts are identified, a regulatory flexibility analysis must be conducted prior to issuing a final rule to determine if small entities should be regulated in a different manner to reduce their economic burden. The Act is designed to identify regulations that will have adverse effects on small entities that are substantially greater than the effects on larger entities, and to require the regulatory agency to consider alternatives that will mitigate these differential impacts. This section analyzes the impacts of the Subtitle D regulation on small governmental entities.

Definitions of Substantial Number, Small Entity, and Significant Impact

The Act and previous regulatory flexibility analyses provide some guidance in developing definitions of what constitutes a substantial number of small entities, what size criteria define a small entity, and what a significant impact is, although it allows EPA to develop a more appropriate definition if necessary. A "substantial number" for the purposes of the Act is typically defined as more than 20 percent of the affected population of small entities, although the absolute number of entities affected may also be considered. The Act provides an initial definition of a small governmental entity as any government with a population of less than 50,000.

To determine how significant the impacts are, a number of measures have been used (e.g., compliance costs as a percentage of production costs; compliance costs as a percentage of sales; capital costs as a proportion of costs available; number/proportion of small entities likely to close). For each measure, a threshold value is assigned; when the threshold is exceeded, this constitutes a "significant impact."

Impact Measures and Results

We adopted the indicator/threshold approach to determine significant impact on small government entities. The indicators selected are those used in the economic impact analysis with the following thresholds:

Indicator	Threshold
Cost as a percentage of expenditures	1%
Cost per household	$220
Cost as a percentage of median household income	1%

The municipal database used in this analysis contains 29,017 entities; 97.6 percent of these entities have a population of 50,000 or smaller. Since such a large proportion of affected entities in our analysis meets the 50,000 population criterion suggested in the Act, and since impacts are greater on entities with less then 5,000 population, an alternative definition of a small entity is appropriate. There are 22,205 entities with populations of 5,000 or less; this represents 77 percent of the total. The proposed regulation will have its most severe impacts on governments serving less than 1,000 people, which include 46 percent of primary local governments. Therefore, an appropriate size definition for small entities for the purpose of a regulatory flexibility analysis could fall somewhere between governments of 5,000 persons and 1,000 persons.

Table VIII-7 summarizes the large number of small jurisdictions that exist, given alternative definitions of small. The table also shows the number of landfills that serve at least one of these "small" jurisdictions.

Table VIII-7

Small Jurisdictions and Landfills That Serve Them

Population Cut-Off	Primary Jurisdictions	Landfills[1]
≤ 1,000	13,211	3,306
≤ 5,000	22,205	4,925
≤ 10,000	24,883	5,317

---

[1] Landfills that serve at least one jurisdiction meeting the size criterion.

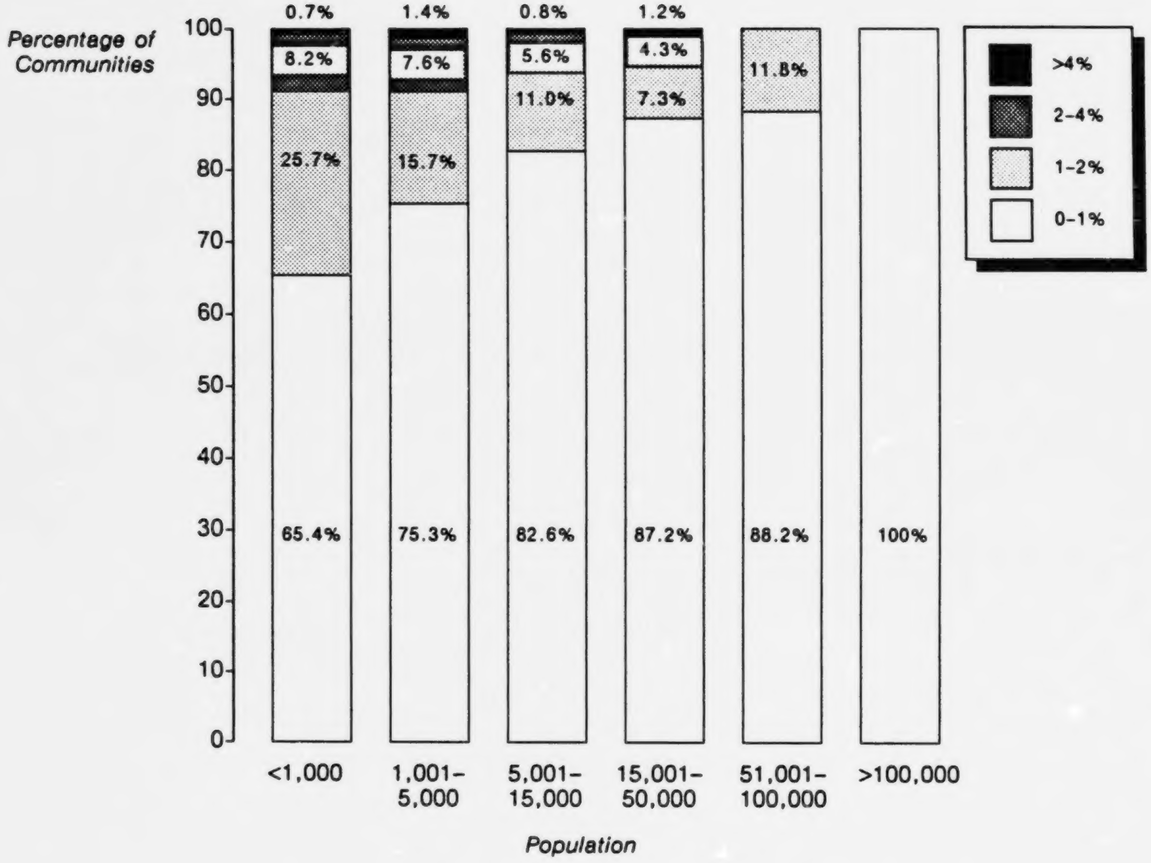
Figures VIII-10 and VIII-11 present data on cost as a percentage of expenditures and cost per household, both broken out by community size, for the final rule assuming use of the reasonable upper-bound cost estimates (40 years of post-closure care and no adjustment for regionalization or expanded use of combustion). (The pattern of impacts is very similar for cost as a percentage of median household income and is not displayed as part of this analysis.) The two indicators show similar patterns of impact, with the greatest impacts falling on communities with populations of 5,000 or fewer; as described earlier in the chapter, most of these are in rural areas. (The final rule exceeds significant impact thresholds for two of the impact indicators: CPE and CPMHI. Only 0.07 percent of communities exceed the threshold for CPMHI, so we focus our attention on the CPE measure as we do in the economic impact analysis.) CPE exceeds 1 percent for 7,822 communities, or 27 percent of all communities in the analysis. Of these 7,822 communities, 6,803 or 87 percent have populations of less than 5,000. CPE exceeds the 1 percent threshold for 31 percent of all communities smaller than 5,000 people and for 35 percent of communities smaller than 1,000 people. Figure VIII-10 clearly shows that the rule imposes different impacts across community size.

The question of the significance of those impacts is less clear, however. First, the final rule imposes less burden on small communities than most of the other alternatives considered. Ground-water monitoring costs are lower under the final rule than under the proposal, for example, and that eases burdens, especially for small landfills where monitoring comprises so much of the cost. Also, the final rule includes less stringent design standards than under Alternative 2 and, in most cases, Alternative 3.

Figure VIII-10

Incremental Cost as Percentage of Expenditures for Final Rule

By Community Size; 40-Year Post-Closure Care

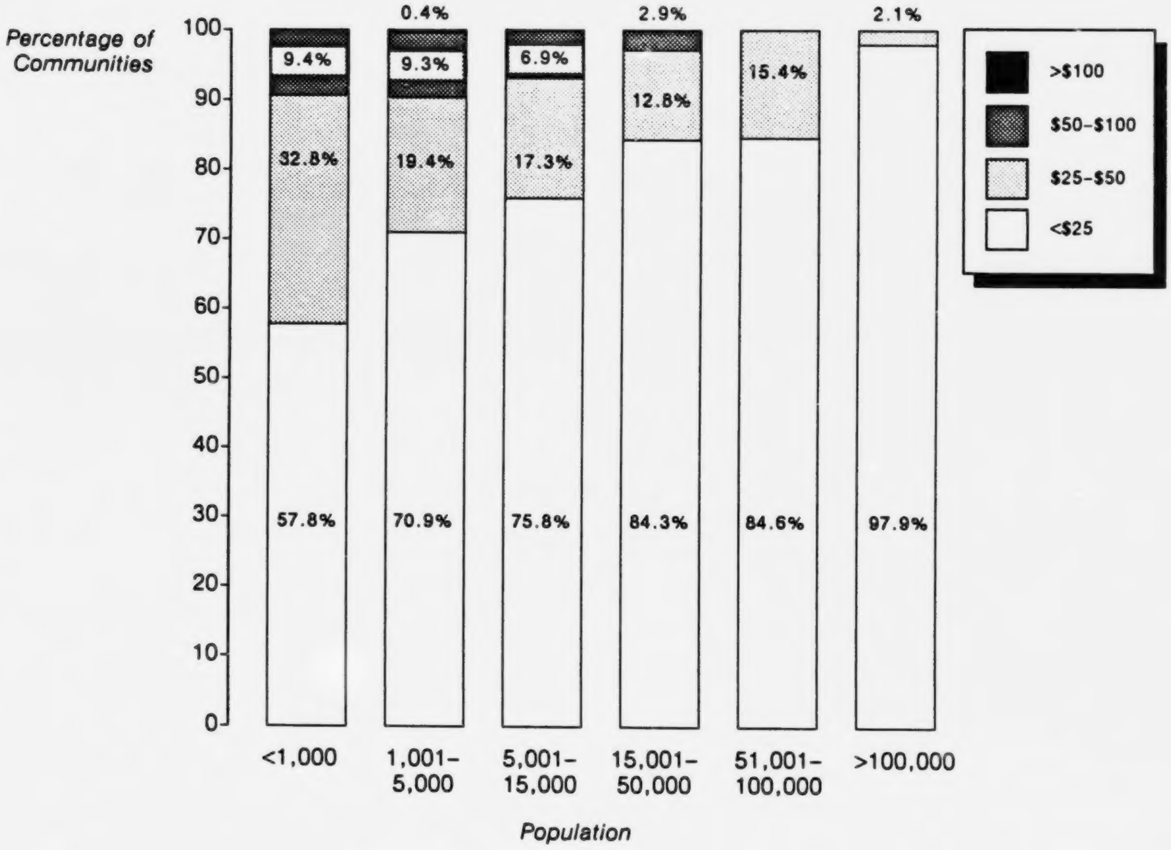


Source: TBS calculations.

Figure VIII-11

Cost per Household per Year for Final Rule

By Community Size; 40-Year Post-Closure Care



Source: TBS calculations.

Table VIII-8 lists the percentage of small communities (under two different definitions) in which CPE exceeds 1 percent. The proposal and Alternatives 2 and 3 all impose greater burdens on small communities. A second factor to consider is the relatively low magnitude of community expenditures for which solid waste management accounts. In Chapter V, we reported that the average expenditure per community on solid waste was about 0.5 percent; while a CPE of 1 percent represents a significant increase, it does not represent a large *absolute* share of the communities' budgets. The majority of the communities with CPE above 1 percent have increases of between 1 and 2 percent as shown in Figure VIII-12. Less than one-third of these communities have increases above 2 percent.

Table VIII-8

Share of Small Communities That Face Significant Impacts[1]

(all options; 40-year post-closure care)

	Population Less Than 1,000	**Population Less Than 5,000**
Final Rule	34.7%	30.6%
Alternative 1 (Proposal)	40.2	34.2
Alternative 2 (Subtitle C)	80.1	71.7
Alternative 3 (Categorical)	47.3	41.2
Alternative 4 (Statutory Minimum)	19.2	16.6

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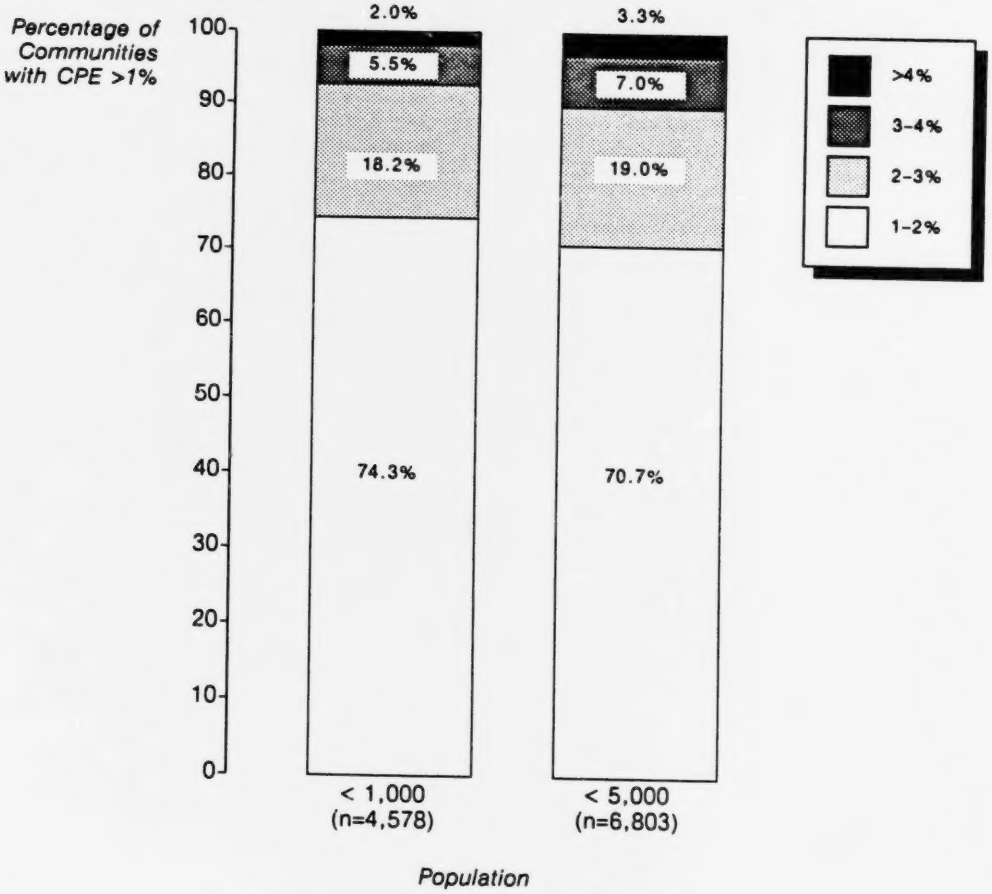
[1] CPE is greater than 1 percent.

Figure VIII-12

Distribution of CPE for Small Communities Facing Significant Increases



Note: Data based on 40-year post-closure care.

Source: TBS calculations.

A third factor that we incorporated into the economic impact analysis was the impact of the reasonable upper-bound cost estimates on which the impacts are based. We are assuming 40 years of post-closure care, no increased diversion of MSW from landfills, and no shifts to larger regional landfills. The example at the end of the economic impact discussion showed how regionalization, at least, would have significant mitigating effects on impacts because regionalization affects small landfills and small communities most directly. Many of these small communities are the ones that will respond to these higher costs by not replacing their existing small landfills, and shifting to larger landfills instead. The analysis showed that after 20 years shifts due to regionalization alone would reduce the number of communities subject to significant impacts by 84 percent. Instead of 27 percent of communities with significant impacts the estimate falls to about 4 percent. We also calculated that if only 10 years of post-closure care were required, the share of communities experiencing severe impacts would fall by 13 percent, from 7,822 communities to 6,835.

Finally, the preamble to the final rule describes several alternatives EPA considered to address the concerns of small communities; some aspects of the alternative that EPA chose are not reflected in our modeling assumptions, and we, therefore, may overstate some of these impacts. EPA considered retaining the 1988 proposal (which we indicated had more severe impacts than the final rule) and considered exempting all small landfills and small communities from some of the requirements; in the final rule EPA does allow approved states to waive requirements at landfills in remote arid areas, and we did not simulate the effect of these exemptions. The exemption was narrowed because EPA wanted to preserve some incentive for better designs at small landfills and to maintain an incentive for regionalization. EPA does exempt all existing landfills from design requirements in the final rule, and we capture that exemption in the modeling. EPA also allows size to be considered in phasing in ground-water monitoring requirements and in setting the post-closure care period. While we modeled phased ground-water monitoring, we imposed monitoring on landfills based on their remaining lives (see Chapter IV), not on size. We also modeled a shorter post-closure period but did not specifically analyze the differential impacts on small communities.

The significance of the impacts on small communities is reduced by several factors: the low magnitude of the increase in CPE for most small communities; the ability of small communities to reduce cost through regionalization over time; the ability of small communities to reduce design costs by locating new landfills at less vulnerable sites; and the opportunity for some small landfills to take advantage of the waiver provision in the final rule. As a result, we conclude that while small communities bear comparatively more of the impact of the final rule, those impacts will not, in the long run, be significant.

One other group of small entities that will be affected by the regulation are the private landfill owners, who own 15 percent of all landfills as reported in the MSWLF survey. For these facilities, we assume that virtually all of them will be able to pass regulatory costs on to the municipalities they serve. This is reasonable given the relatively tight supply of landfills in most parts of the country. Also, because the severe impact threshold was never exceeded for households under the final rule, even a 100 percent pass-through would not significantly affect residents.

In some cases that are very difficult to predict, however, private owners may not be able to pass costs on and would have to absorb the incremental costs. For the purposes of the Regulatory Flexibility Analysis, we should evaluate whether small owners would be affected more severely than large ones. Because we do not have more detailed information on private landfill owners, we assume that each small private landfill operates as an independent business (i.e., it is run by a small entity).

While this assumption is not entirely correct, it is a good proxy for the impact on private owners, since large corporate owners of landfills may not continue to operate a landfill that is losing money, just as a small business would cease operations if it were losing money. It is possible, however, that small or large business landfill owners who also *collect* MSW may be willing to absorb landfill compliance costs against collection profits and would continue to evaluate their entire operations as an independent entity. In addition, for the purposes of the Regulatory Flexibility Act, it will be extremely difficult (based on available data) to distinguish between small businesses and businesses controlled by large corporations, even though both entities run small landfills. For those reasons, we treat all small landfills as small entities for this analysis.

Our analysis of the final rule incremental costs faced by private landfills indicates that these costs are substantial (the median increase above baseline costs as reported in the survey is 98 percent). Table VIII-9 indicates, however, that these increases are not significantly higher for small landfills than they are for large ones. The median increase for all private landfills is 98 percent; the median for the smallest of these landfills is 88 percent, so it is not clear from the data that small private landfills are affected any more severely than government-owned landfills.

Table VIII-9

Incremental Costs of the Final Rule to Private Landfill Owners

(percentage increase above baseline cost reported in the survey[1]; 40-year post-closure care)

Size (TPD)	Median Percentage Increase
10	88%
25	97
75	122
175	85
375	98
750	59
1,500	NA
Overall	98

---

[1] These baseline costs are probably understated as described in Chapter VII.

NA = Not available.

Conclusion

This analysis of small entity impacts indicates that there are clearly differential economic impacts on a substantial number of small governments. Although the impacts are more severe than those on large communities as measured by the indicators, it is not clear that these impacts are significant and are incurred by a substantial number of small communities. The chapter describes how the cost assumptions overstate impacts because they assume 40 years of post-closure care, do not allow for regionalization or increased waste diversion, and do not consider better locations and lower design costs for future landfills. We did analyze the impact of shortening post-closure care to 10 years (which reduced by 13 percent the share of communities facing significant impacts) and of regionalization (which reduced the severely affected communities by 84 percent after 20 years). Given these findings and additional factors such as increased waste combustion, increased recycling and source reduction, and exemptions for small communities, we conclude that significant impacts will not fall on a substantial number of small communities. We also believe that most small private landfill owners will be able to pass costs on to customers. For those who cannot, the data suggest that while costs may be significant, they do not fall disproportionately on small private landfills.

IX. Results: Risk and Resource Damage

In this chapter we present the results of the health-risk and resource-damage analysis for the baseline, the final rule, and each regulatory alternative. The risk results are discussed first, followed by the resource damage results.

Health Risk

As discussed in Chapter VI, we analyzed 300-year average risks for the most-exposed individual and for the population at large. In our discussion of average individual risk results below, we define, *for this analysis only*, the risk categories as follows:

- High -- greater than 10^{-5}
- Moderate – 10^{-6} to 10^{-5}
- Low – 10^{-8} to 10^{-6}
- Very low – less than 10^{-8}

We emphasize that the risk categories are defined in this way for this analysis only, and do not represent Agency policy regarding high, moderate, low, and very low risk to individuals.

Baseline Risk

The baseline case assumes that all landfills are new, unlined, and, when closed, will be covered with topsoil and vegetation. In this section, we discuss the model outputs in terms of the overall distribution of risks and the factors that influence the magnitude of risk to human health.

Distribution of Risk

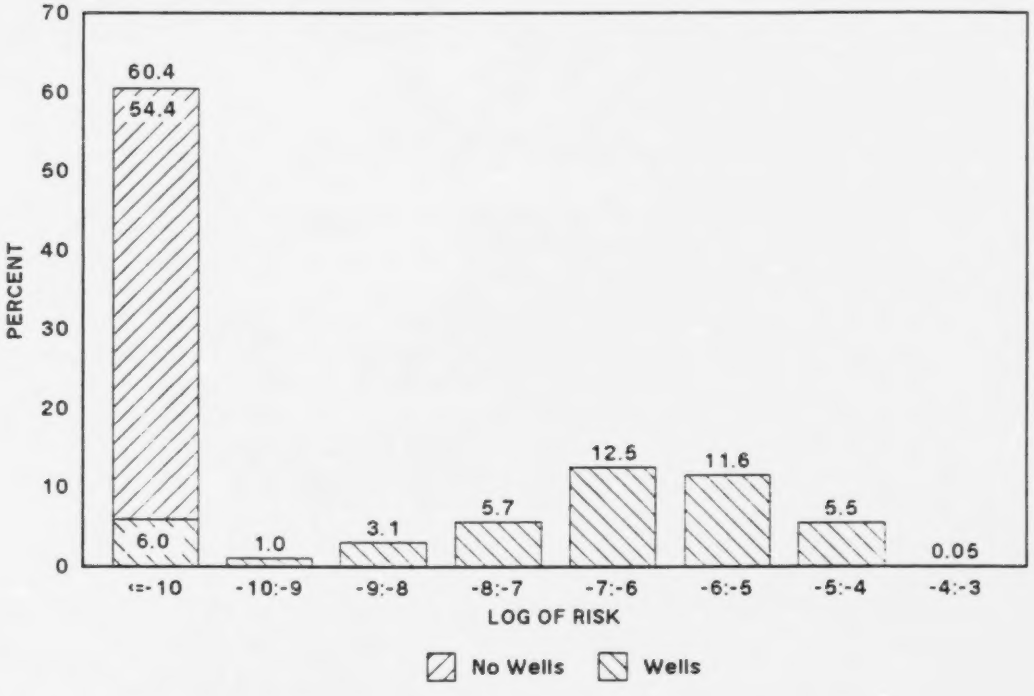
Figure IX-1 presents a frequency distribution of 300-year risk to maximum exposed individuals (MEIs) across all units. These results have been weighted by taking into account the relative frequency of net infiltration, water-table depth, hydrogeologic flow settings, landfill size, and distance to the nearest ground-water well. The maximum exposed individual is a person exposed at the well closest to the landfill (as identified in the MSWLF survey). The height of each bar in Figure IX-1 indicates the proportion of landfills modeled where risks fall into a given range (e.g., 10^{-7}:10^{-6} means that risk is greater than 10^{-7} and less than or equal to 10^{-6}). A risk of 10^{-6} means that an exposed individual would bear a 1 in 1,000,000 increased chance of contracting cancer in his lifetime as a result of the exposure.

The figure shows that risks range from zero to about 1 x 10^{-4}. Over 54 percent of landfills have no nearby wells (i.e., no wells within one mile of the facility boundary, as reported in the MSWLF survey) and hence have no modeled human health risk. Another 6 percent have nearby wells but have no risk because constituents do not reach the wells within the modeling period. Risks are low (10^{-6} to 10^{-8}) or extremely low (less than 10^{-8}) for a total of 82.8 percent of landfills. Of the remainder, 11.6 percent have moderate risk (i.e., fall into the 10^{-6} to 10^{-5} range), 5.5 percent have high risk (10^{-5} to 10^{-4}), and 0.05 percent exceed 10^{-4}. The principal constituents contributing to risk are vinyl chloride, 1,1,2,2-tetrachloroethane, and dichloromethane (methylene chloride). While arsenic is the primary constituent of concern in some environmental settings, risks for these settings rarely exceed 10^{-6}.

Figure IX-1

Distribution of 300-Year Average MEI Risk[1]

Baseline



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[1] The MEI or maximum exposed individual is a person exposed at the well closest to the landfills (as identified in the MSWLF survey).

Another way to portray baseline risk is in terms of a cumulative distribution, as presented in Figures IX-2 and IX-3. The y-coordinate for any point on these graphs corresponds to the percentage of landfills with risks exceeding the point's x-coordinate. Figure IX-2, the cumulative distribution for all landfills, shows that the median risk is less than 10^{-10}. This makes sense, because over half the landfills have no nearby wells and therefore no risk. The figure also shows that less than 20 percent of landfills have a risk greater than 10^{-6}.

When we consider only landfills that have nearby wells (Figure IX-3), a different picture emerges. For this subgroup of the population, the median risk is about 4.3×10^{-7}. In addition, nearly 40 percent of the landfills have moderate or high risks. This distribution has important implications because as more landfills shift into this subgroup (the result of new wells being drilled or new landfills being located near existing wells), the risks for *all* landfills will tend to approach this distribution.

We predict about 0.077 expected cancer cases per year over the 300-year modeling period.

Factors Affecting Baseline Risk

Risk from municipal solid waste landfills varies depending upon the size of the landfill, the distance to the nearest ground-water well, and the environmental setting, including net infiltration rates, water-table depth, and flow field type. In this section of the report, we examine how these factors affect risk. In order to focus on the impact of each factor, the results are presented in unweighted terms (i.e., each landfill size/environmental setting/well distance combination is weighted equally).

Facility Size. As Figure IX-4 shows, facility size is an important determinant of risk. Higher levels of contamination, and hence higher risks, are associated with larger facilities that have a greater mass of waste. In 10-TPD facilities, about 33 percent of the scenarios have risks that exceed 10^{-6}. This value increases to over 55 percent for 175 TPD and nearly 64 percent for 750 TPD. Eight percent of the 10-TPD scenarios have risks exceeding 10^{-5}, 22 percent of the 175-TPD scenarios exceed 10^{-5}, and nearly 30 percent of the 750-TPD scenarios exceed 10^{-5}. Since only 5.5 percent of landfills fall into the 750-TPD category, the impact of these landfills on the overall distribution is small. The small (10-TPD) facilities account for almost 62 percent of landfills; their lower risk profile tends to bias the overall distribution toward the low side of the risk spectrum.

FIGURE IX-2

CUMULATIVE FREQUENCY OF AVERAGE RISK

BASELINE – ALL LANDFILLS

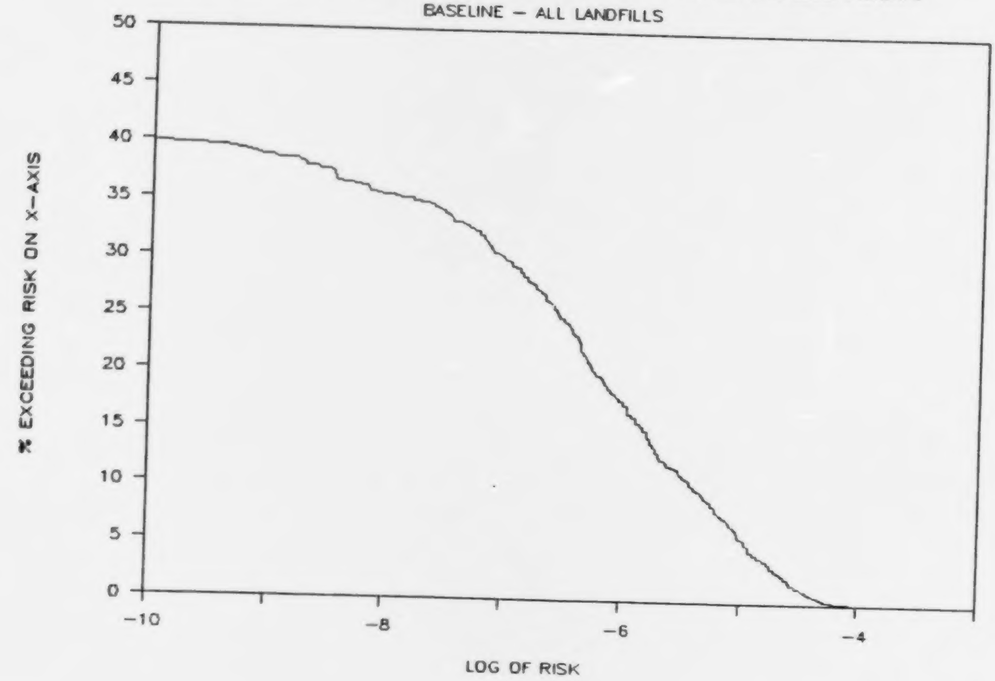


FIGURE IX-3

CUMULATIVE FREQUENCY OF AVERAGE RISK

BASELINE – ONLY LANDFILLS WITH WELLS

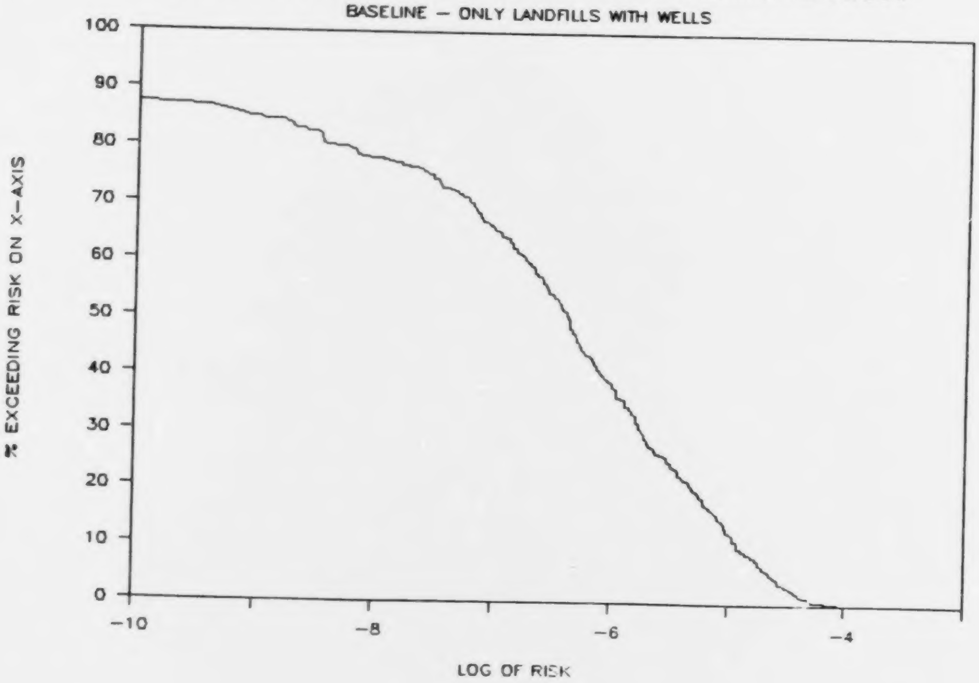
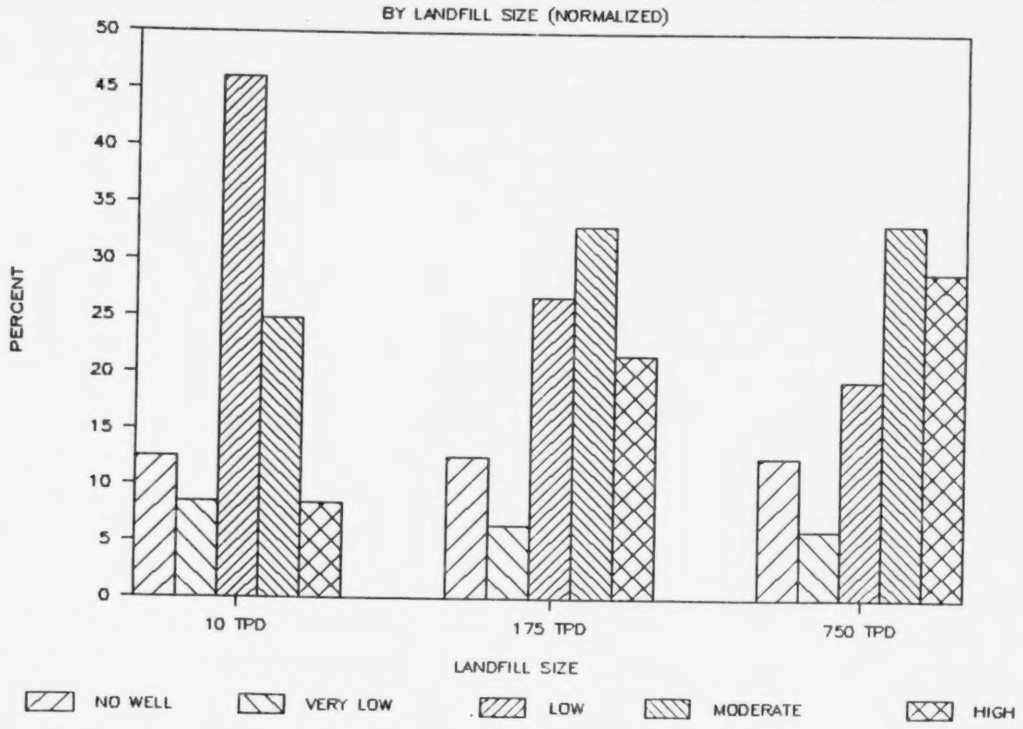


FIGURE IX-4

DISTRIBUTION OF AVERAGE RISK -- BASELINE



Note: These results are not weighted to represent the frequency of occurrence of each landfill size, environmental setting, or well distance.

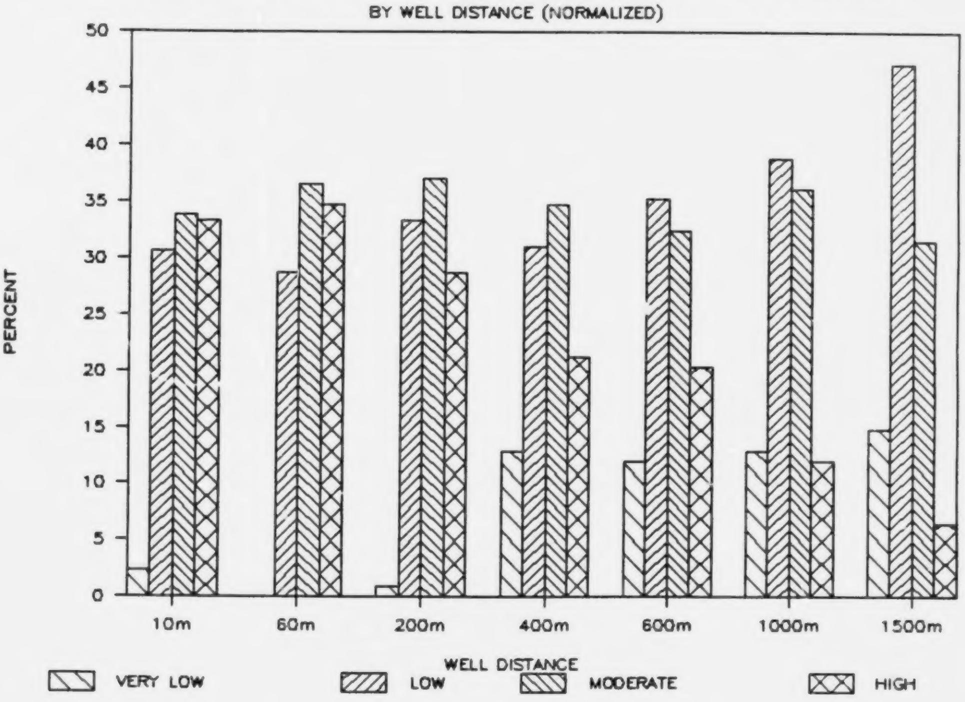
Distance to Nearest Well. All other factors held constant, risk decreases with distance from the facility (Figure IX-5). Contaminant concentrations diminish over distance due to degradation, dispersion, and attenuation. At 10 meters, over 67 percent of the settings have risks exceeding 10^{-6}; 34 percent exceed 10^{-5}. On the other hand, only about 2 percent have very low risk. At 1,500 meters, only 40 percent of the scenarios have risks exceeding 10^{-6} and almost 15 percent have very low risk (less than or equal to 10^{-8}).

The influence of well distance on the overall risk distribution depends on the relative weights assigned to each distance. The fact that over 54 percent of landfills have no wells (within one mile) sharply diminishes overall risk. Only 3 percent of the landfills have their closest wells within 35 meters; 12 percent have wells between 35 meters and 300 meters; 12 percent have a well in the 300- to 500-meter range; and the remaining 20 percent have closest wells between 500 and 1,600 meters. While the closest wells have the greatest risk, their occurrence is relatively rare, and their impact on the overall risk distribution is consequently small.

FIGURE IX-5

DISTRIBUTION OF AVERAGE RISK - BASELINE

BY WELL DISTANCE (NORMALIZED)

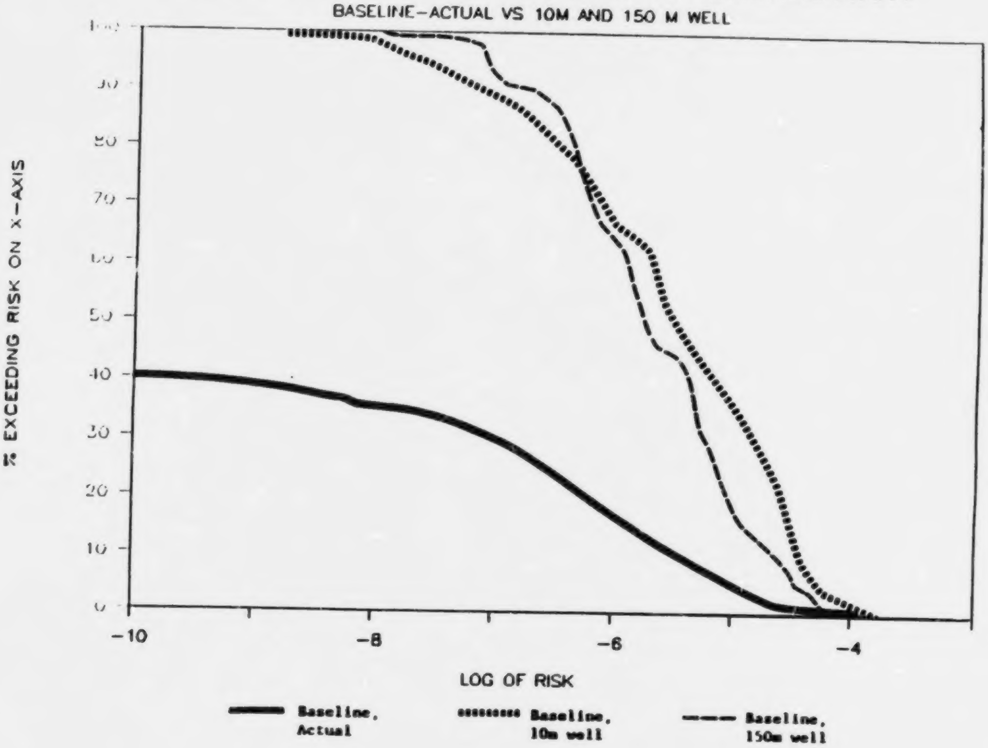


Note: These results are not weighted to represent the frequency of occurrence of each landfill size, environmental setting, or well distance.

In Figure IX-6, we compare the cumulative frequency of average risk assuming that exposure occurs: (1) at the actual exposure well distribution; (2) at the unit boundary of all landfills (which we simulate using the 10-meter well distance); and (3) at a distance of 150 meters from the unit boundary. Figure IX-6 reveals that the well distance distribution has a profound effect on risk. While about 5.6 percent of the landfills have risks exceeding 10^{-5} at the actual exposure points, about 34 percent exceed this risk level at the unit boundary and 20 percent do so at the 150-meter distance. Less than one-fifth of the landfills have risks higher than 10^{-6} when we consider the actual well distribution, compared to about 65 percent at both the unit boundary and the 150-meter distance.

FIGURE IX-6

CUMULATIVE FREQUENCY OF AVERAGE RISK

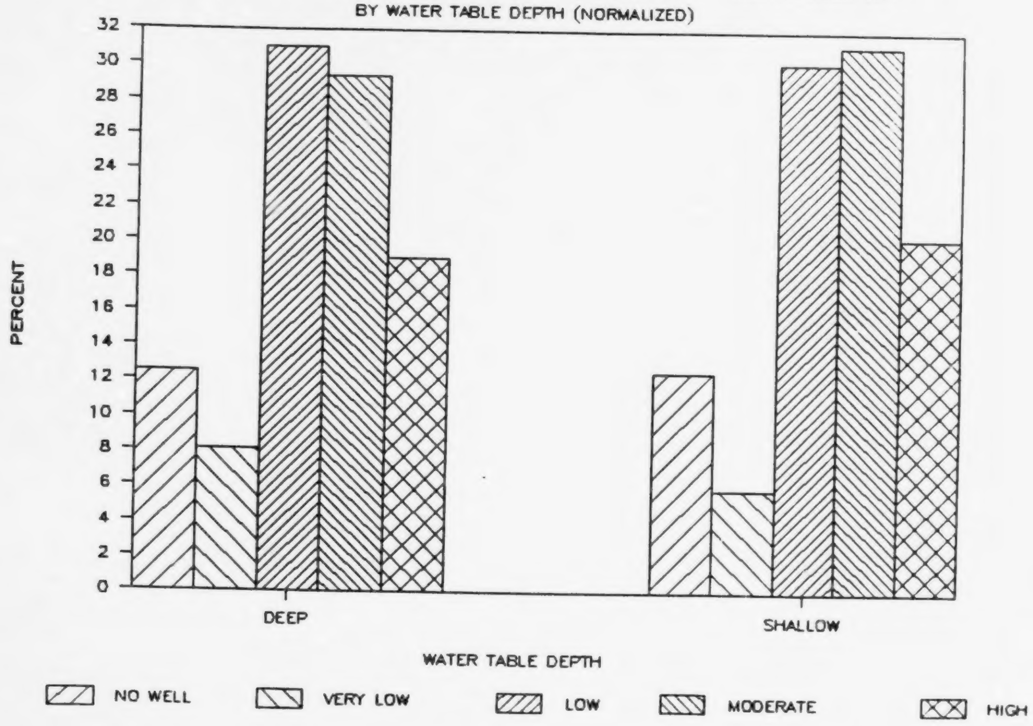


Water Table Depth. Figure IX-7 shows a very slight difference between risk distributions in shallow and deep water-table settings. The deep settings have about 3 percent fewer scenarios in the moderate- and high-risk ranges. This slight shift toward lower risk in deep water-table settings results from longer unsaturated zone travel times, which allow for more pollutant degradation prior to release to the aquifer. This effect is probably more pronounced in arid regions, where the differences in unsaturated zone travel times between the deep and shallow settings are greater.

Results of a regression analysis of the data confirm that water-table depth has little effect on overall risk. The analysis shows that unsaturated-zone travel time explains only 3 percent ($R^2 = 0.03$) of the variation in risk.

FIGURE IX-7

DISTRIBUTION OF AVERAGE RISK - BASELINE



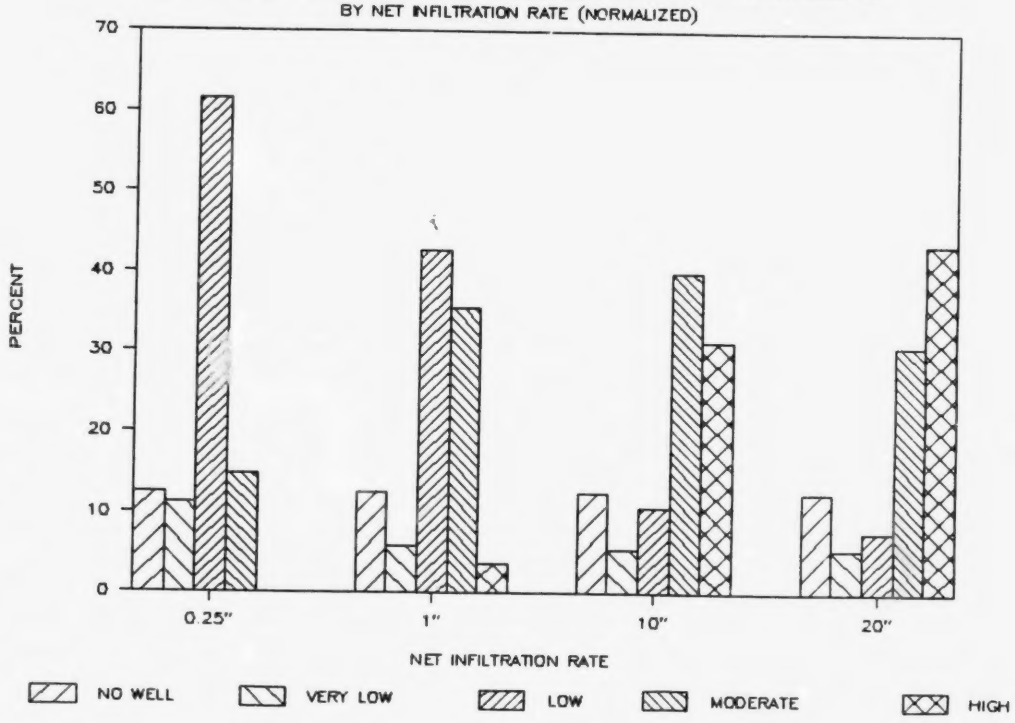
Note: These results are not weighted to represent the frequency of occurrence of each landfill size, environmental setting, or well distance.

Net Infiltration. Wetter climates are associated with higher release volumes and consequently greater risks, as shown by Figure IX-8. For a net infiltration rate of 0.25 inches, the risk exceeds 10^{-6} in only 15 percent of the scenarios and never exceeds 10^{-5}. In the wettest setting (20 inches), the risk is moderate (10^{-6}:10^{-5}) in over 30 percent of the scenarios and high (greater than 10^{-5}) in over 42 percent of the scenarios. For intermediate values of net infiltration, risk increases with infiltration. The high-risk profile associated with the 20-inch infiltration region is mitigated by its relatively low frequency (about 12 percent). Over two-thirds of the landfills are split about equally between the 0.25- and 10-inch settings, which have the greatest effect on the overall risk distribution. The results of our regression analysis show a moderate relationship between net infiltration and log of risk ($R^2 = 0.35$). However, the fact that net infiltration is a categorical variable with only four values limits both the usefulness of least-squares regression as an analytical tool and the validity of the results.

FIGURE IX-8

DISTRIBUTION OF AVERAGE RISK - BASELINE

BY NET INFILTRATION RATE (NORMALIZED)



Note: These results are not weighted to represent the frequency of occurrence of each landfill size, environmental setting, or well distance.

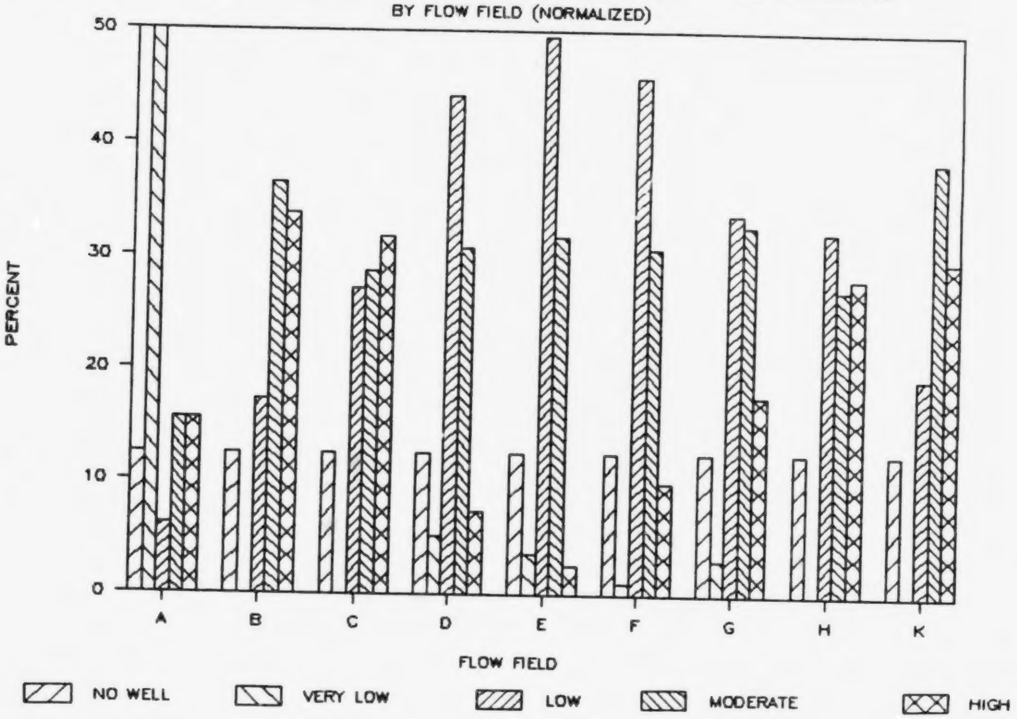
Flow Fields. Hydrogeologic characteristics of the aquifer exert a strong influence on risk. Aquifer properties affect the extent of dilution of the leachate, and the retardation and degradation of pollutants.

Of the five single-layer flow fields simulated (i.e., water-table aquifers defined by flow fields A through E), the slowest (flow field A, 1 m/yr velocity) and the fastest (flow fields D and E, velocities of 1,000 m/yr and 10,000 m/yr, respectively) have lower risk profiles than flow fields B and C (10 m/yr and 100 m/yr, respectively) (see Figure IX-9). The slow velocity of flow field A prevents pollutants from reaching the more-distant wells during the modeling period, and allows for considerable pollutant degradation before breakthrough at nearby wells. In the high-velocity flow fields, the large volumes of water flowing through the aquifer afford more dilution of the leachate. Flow fields B and C have higher risk profiles because they neither allow for much degradation nor provide for much dilution or dispersion of pollutants.

FIGURE IX-9

DISTRIBUTION OF AVERAGE RISK - BASELINE

BY FLOW FIELD (NORMALIZED)



Note: These results are not weighted to represent the frequency of occurrence of each landfill size, environmental setting, or well distance.

The leaky confined aquifers (flow fields F and K) have somewhat lower risks than the corresponding single-layer flow fields with similar characteristics (B and C). The risks are reduced by the additional degradation and attenuation that occur during seepage through the confining layer.

Finally, the double-layer flow fields (G and H) have risks somewhat lower than the analogous single-layer flow fields (B and C). The double-layer flow fields each have a 10 m/yr layer and a 100 m/yr layer, but differ in geometry and layer thickness. These flow fields have slightly higher rates of aquifer flux than do either flow fields B or C, and thus have slightly more dilution capacity. Also, because we estimate concentrations for the lower layers in flow fields G and H, attenuation and degradation in the upper layer act to reduce risks.

Constituent Concentrations in Leachate. Our simulation of baseline risk was based on the median levels of contaminants observed in leachate at actual landfills. This means that half of the landfills would be expected to have leachate with higher contaminant levels than what we simulated, and half would have lower levels. To place an approximate upper bound on contaminant concentrations and the associated risk levels, we determined the revised 90th percentile level of each constituent of concern (see Chapter VI) and compared it to the median level simulated in the model. Table IX-1 shows the ratios of the revised 90th percentile to the median level for each of the six constituents that we modeled for human health risk.[1]

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[1] As discussed in Chapter VI, the median values used as a basis for the risk and resource damage analysis were based on the original leachate database, which covered 44 operating landfills. We later calculated new medians and 90th percentiles based on a revised database comprising 99 landfills. The ratios presented in Table IX-1 are the ratios of the revised 90th percentiles to the median concentrations used in the model (not the revised median concentrations). The new median values for the eight constituents of concern were not significantly different from those simulated in the model.

Table IX-1

Ratios of Revised 90th Percentiles to Median Levels Simulated in the Model for Constituents of Concern

Constituent	Ratio
Dichloromethane	6.6
Vinyl Chloride	4.7
Arsenic	3.0
Carbon Tetrachloride	2.0
1,1,2,2-Tetrachloroethane	2.0
Antimony	1.4

The revised 90th percentile concentrations of the three principal constituents contributing to risk (vinyl chloride, dichloromethane, and 1,1,2,2,-tetrachloroethane) are two to about seven times higher than the median levels modeled. Because all of these constituents are carcinogens, and because we assumed that cancer risk varies linearly with dose, we believe that the risk associated with the revised 90th percentile levels would be less than one order of magnitude higher than that simulated in the model. This indicates that at 10 percent of the landfills, risks from individual constituents would be up to 10 times higher than our results indicate.

Summary. About 17 percent of landfills have risks exceeding 10^{-6}, a proportion that could increase if new wells are drilled in the vicinity of landfills or if new landfills are sited near existing wells. When we consider the factors that influence risk, we find that infiltration rate, facility size, distance from the facility, and aquifer characteristics are strong determinants of risk. No *single* factor is responsible for most of the variability, however; instead, there is a complex interaction among factors. Therefore, it is difficult to reliably predict risk without accounting for the interaction among the factors we have modeled. Moreover, because our model employs a number of simplifying assumptions, there are many additional interactions not accounted for here.

Risk Reduction – Final Rule

In this section we discuss the effectiveness of the final rule in reducing risks to human health. Unlike the cost analysis, which simulates costs for two different post-closure care periods (10 years and 40 years), the risk analysis assumes that the post-closure care period is 30 years. As discussed in Chapter VI, we assumed that 27 percent of all landfills would be required to comply with a uniform design standard under the final rule, and the other

73 percent with an MCL-based performance standard. We estimate that, under the final rule, 56 percent of landfills would be unlined with vegetative covers; less than 1 percent would be unlined with synthetic covers; and the remaining 43 percent would have composite or synthetic liners and synthetic covers.

Figure IX-10 illustrates the distribution of 300-year average individual risk for both the baseline and the final rule. Under the final rule, about 0.6 percent of landfill have risks exceeding 10^{-5}, compared to 5.6 percent in the baseline. About 12.4 percent of landfills have moderate risk (between 10^{-6} and 10^{-5}), compared to 11.6 percent in the baseline. The final rule has more low-risk landfills (i.e., between 10^{-8} and 10^{-6}) than the baseline (22.5 percent compared to 18.2 percent) and about the same number of landfills with very low or no risks.

Figure IX-11 shows the risk reduction achieved under the final rule for all combinations of landfill size, environmental setting, flow field, and well distance. The points plotted in this figure do not reflect the frequency of occurrence of each combination. The x-axis represents the log of 300-year average individual risk in the baseline, and the y-axis is the log of risk under the final rule. If the point lies on the equal-risk line (the solid line extending diagonally across the graph), risk under the final rule is equal to baseline risk (i.e., the regulation does not reduce risk). If the point lies below the equal-risk line, risk is reduced by the final rule.

Figure IX-11 shows numerous points on the equal-risk line (i.e., scenarios for which the final rule does not reduce risk). These are associated with landfills where the MCL-based performance standard is not exceeded in the baseline. Most of the scenarios with baseline risks higher than 10^{-6} show some risk reduction under the final rule. In most cases, these risks are reduced by less than a factor of 10.

Population risks are 0.025 cancer cases per year over the 300-year modeling period, down from a baseline of 0.077 cancer cases per year.

FIGURE IX-10
DISTRIBUTION OF AVERAGE RISK

FINAL RULE AND BASELINE

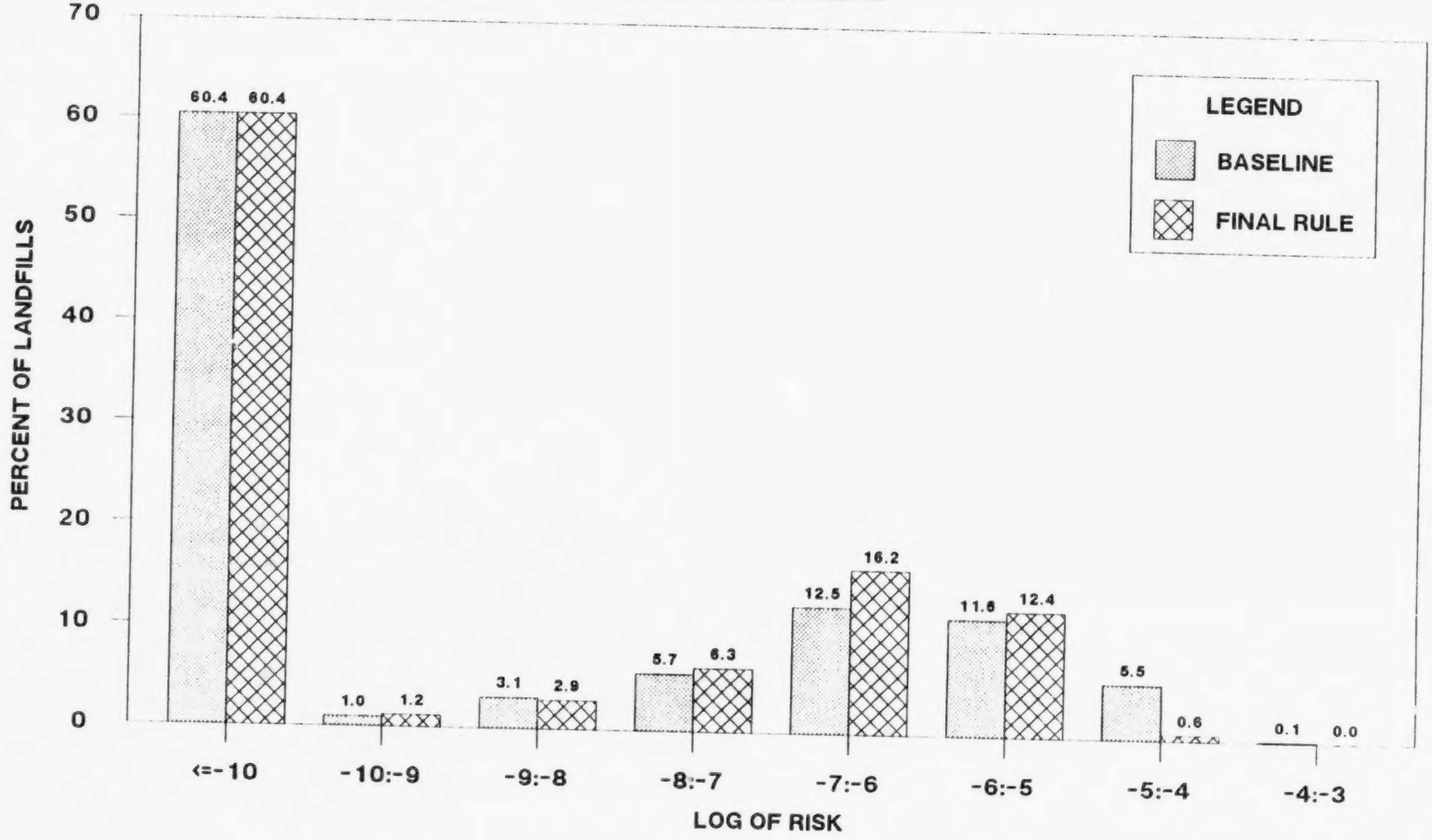
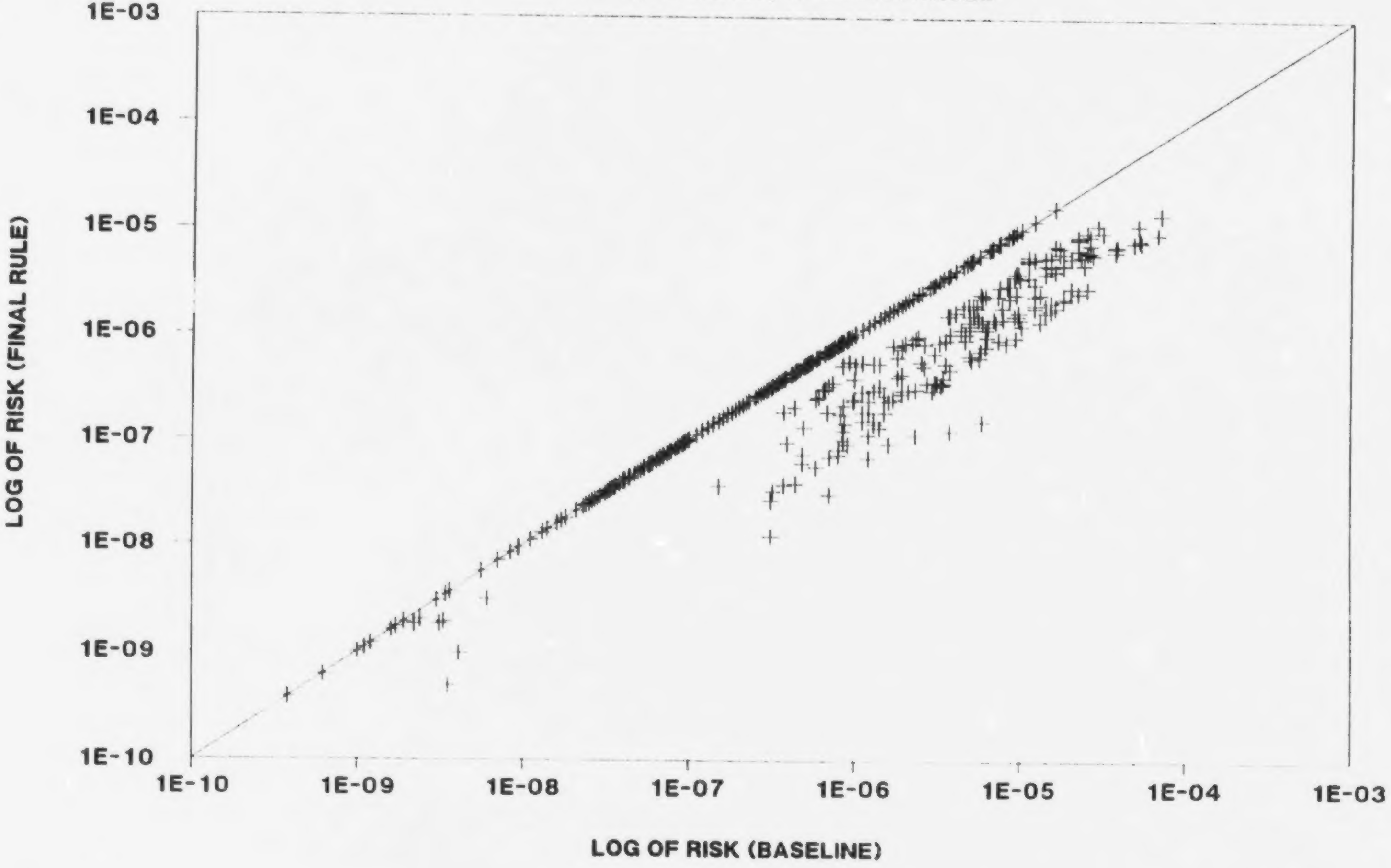


FIGURE IX-11
BASELINE VS. FINAL RULE ALTERNATIVE

ALL SIZES, SETTINGS, WELL DISTANCES



If the post-closure care period were extended to 40 years rather than the 30 years simulated by the model, risks would be somewhat lower than those reported above. For landfills equipped with liners and covers, the additional 10 years to containment would allow more time for degradation of constituents prior to release. However, vinyl chloride is the only constituent of concern that degrades quickly enough for this additional delay to be an important factor. The half-life of vinyl chloride is about 10 years, indicating that an additional 10-year delay could cause vinyl chloride concentrations in leachate to halve. Whether this would cause risks to be significantly lower than those reported above depends on the extent to which vinyl chloride contributes to the risks under the longer care period, which was not modeled.

If the post-closure care period were shortened to 10 years, risks would be higher than those simulated by the model. For landfills designed with liners and covers, there could be up to 20 years less containment prior to leachate release, compared to the model runs. With 20 years less containment, vinyl chloride concentrations could be four times higher than those simulated by the model. Again, it is not possible to quantify the extent to which risk would increase without performing new model runs. Finally, for landfills without liners, the duration of the post-closure care period is not a significant factor.

Risk Reduction – Alternative 1

In this section we discuss the effectiveness of the 1988 proposal in reducing risk to human health. As discussed in Chapter VI, we simulated risk for each scenario using three designs – unlined with vegetative cover, unlined with synthetic cover, and synthetic-lined with synthetic cover – and assigned all landfills to the least stringent design capable of meeting the design goal. Under this alternative, states could choose a design goal anywhere in the protective range of 10^{-4} to 10^{-7}. We assumed that the design goal was set to require average lifetime risk at the compliance point to be 10^{-5} or less. To the extent that states set the goal at a level other than 10^{-5}, the benefits from the rule will differ from those presented here. The compliance point was set at the unit boundary, which we simulated using the 10-meter well distance.

As with the final rule, we simulated 30 years of post-closure care. Modeled post-closure care activities include maintenance of the cover, leachate collection (for designs with an LCS), ground-water monitoring, and corrective action. Corrective action is triggered at the 10^{-5} level.

We estimate that for about 61 percent of all landfills, vegetative covers alone are sufficient to meet a 10^{-5} risk-based design goal at the unit boundary. Synthetic covers are sufficient for 11 percent of the landfills, while synthetic liners plus LCS and synthetic covers are needed at the remaining 28 percent. None of the landfills trigger corrective action under Alternative 1.

Figure IX-12 shows the distribution of 300-year average MEI risk for both the baseline and Alternative 1. About 0.5 percent of the landfills have risks exceeding 10^{-5} under this alternative, compared with 5.6 percent in the baseline. About 35 percent of the landfills have risks between 10^{-8} and 10^{-5} under the 1988 proposal, which is slightly higher than it is in the baseline (30 percent); this reflects a shift in risk from the high to the moderate and low risk categories. The percent of landfills with risks below 10^{-8} is about equal for both the baseline and Alternative 1.

Figure IX-13 shows the risk reduction achieved under Alternative 1 for all combinations of landfill size, environmental setting, flow field, and well distance. As in all graphs of this type, the points do not reflect the frequency of occurrence of each combination and do not include scenarios with baseline risk below 10^{-10}. The figure shows numerous points on the equal-risk line (i.e., scenarios for which the 1988 proposal does not reduce risk). These are associated with landfills that are already in compliance with the performance standard in the baseline. Nearly all scenarios with baseline risks higher than 10^{-5} show some risk reduction under Alternative 1; for these scenarios, risk is usually reduced by less than an order of magnitude, but is virtually always reduced to less than 10^{-5}. Figure IX-13 also shows variable amounts of risk reduction for several scenarios with baseline risk below 10^{-5}. Some of these points represent exposure distances of 60 meters or more at landfills that must implement more stringent containment to meet the performance standard at the unit boundary. Others correspond to landfills in double-layer flow fields G and H, which are assigned more stringent designs or corrective action based on pollutant levels in the upper-aquifer monitoring wells rather than the lower-aquifer exposure wells.

Population risks are 0.028 cancer cases per year (over the 300-year modeling period), down from a baseline of 0.077 cases per year.

FIGURE IX-12
DISTRIBUTION OF AVERAGE RISK

ALTERNATIVE 1 AND BASELINE

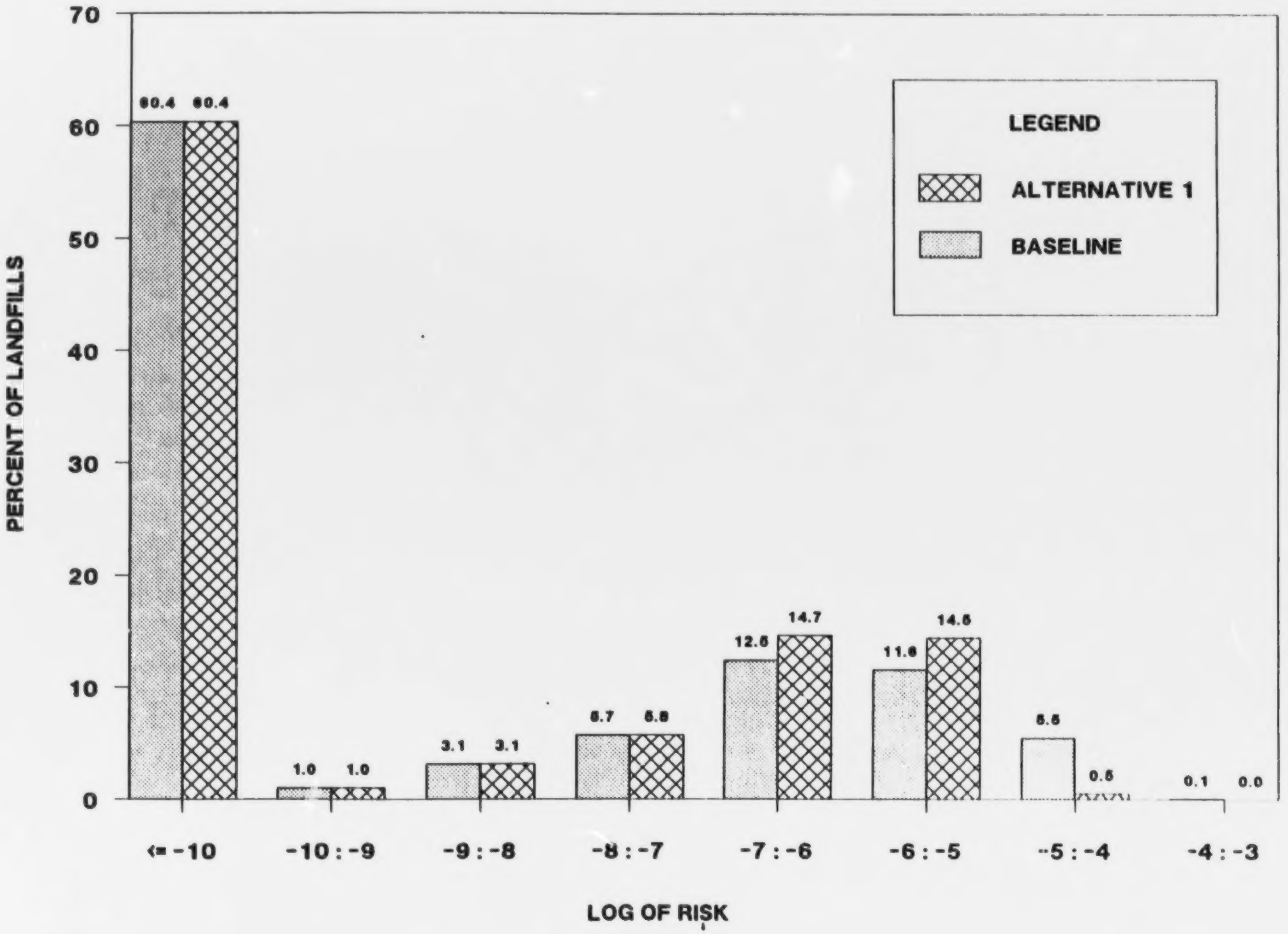
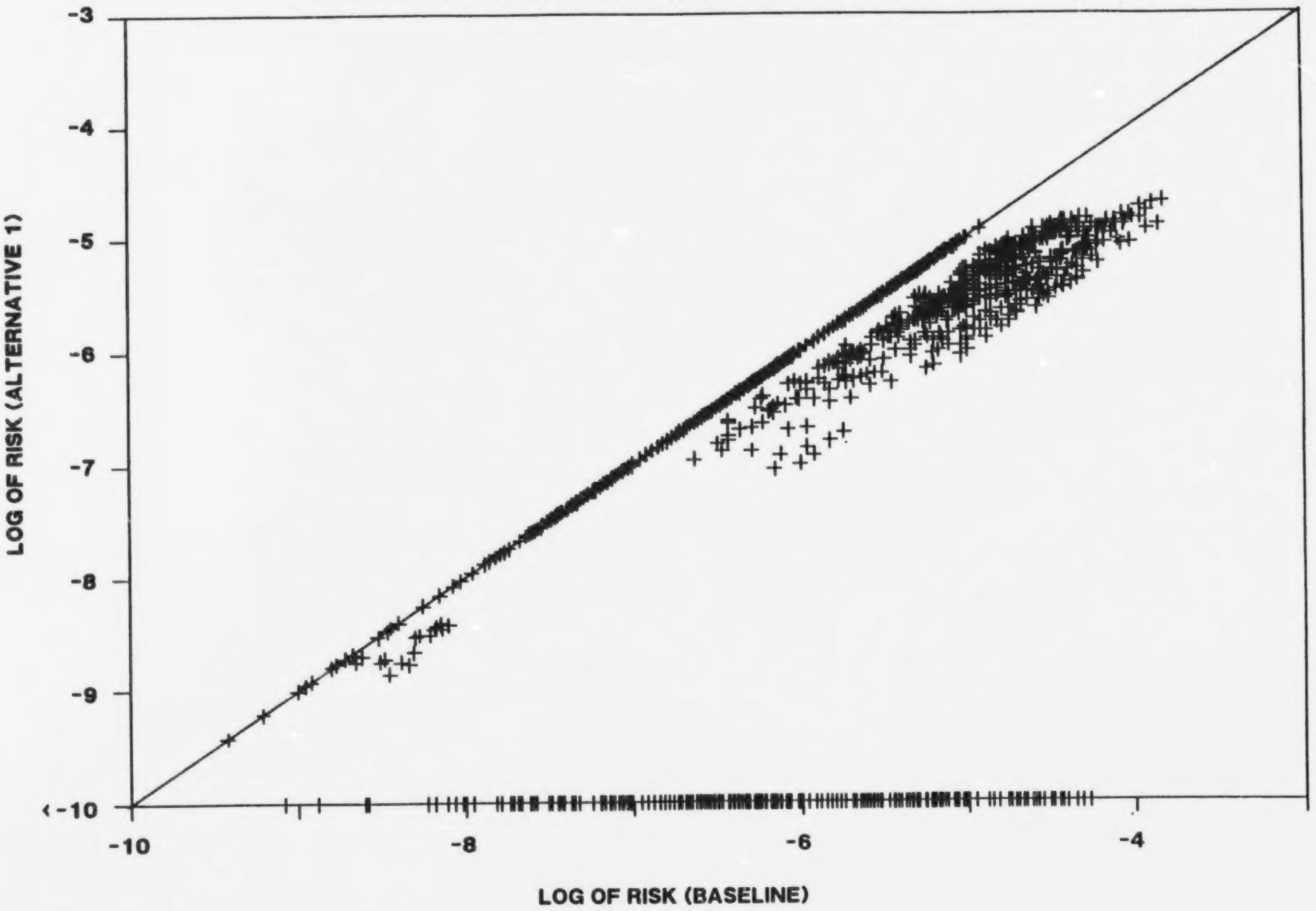


FIGURE IX-13

BASELINE VS. ALTERNATIVE 1 AVERAGE RISK

ALL SIZES, SETTINGS, WELL DISTANCES



These results reflect 30 years of post-closure care. If the post-closure care period were extended to 40 years, risks would be somewhat lower than those simulated by the model. As with the final rule, we are unable to quantify the extent to which risks would decrease with longer post-closure care periods. For landfills with liners and covers, the major effect would be to increase the delay in releases, which would allow more time for vinyl chloride to degrade prior to release. There would be no increase in the percentage of landfills that trigger corrective action, as none of the landfills trigger corrective action in our modeling of this alternative. As discussed in Chapter VI, this is because landfills that exceed a 10^{-5} risk level with only vegetative or synthetic covers are assumed to implement a more stringent design rather than trigger corrective action; and landfills designed with liners and covers do not trigger corrective action anytime before the end of the post-closure care period.

Risk Reduction – Alternative 2

Of all the regulatory options being considered, Alternative 2 has the most stringent containment, leachate collection, and cover requirements: a synthetic/composite liner (two synthetic membranes overlying three feet of clay), two leachate collection systems, and a composite cover (synthetic membrane overlying clay). These requirements are imposed uniformly on new landfills.

As with the final rule, we simulated 30 years of post-closure care. Modeled post-closure care activities include maintenance of the cover, leachate collection (for designs with an LCS), ground-water monitoring, and corrective action. None of the landfills trigger corrective action in our modeling of this alternative.

Figure IX-14 shows the distribution of 300-year average MEI risk for both the baseline and Alternative 2. About 0.3 percent of the landfills have risk greater than 10^{-5} at exposure wells under Alternative 2, compared with 5.6 percent in the baseline; about 7.6 percent have moderate risk (10^{-6} to 10^{-5}), compared with 11.6 percent in the baseline; 14.1 percent have low risk (10^{-8} to 10^{-6}); and the remaining 78 percent have very low or no risk.

Figure IX-15 shows the risk reduction achieved under Alternative 2 for all combinations of landfill size, environmental setting, flow field, and well distance. The points plotted in this figure do not reflect the frequency of occurrence of each combination and do not include scenarios with baseline risk below 10^{-10}. The figure shows that in most cases risk is reduced by a factor of 2 to 20 under Alternative 2. There are, however, several combinations for which the risk reduction is substantially greater.

FIGURE IX-14
DISTRIBUTION OF AVERAGE RISK

ALTERNATIVE 2 AND BASELINE

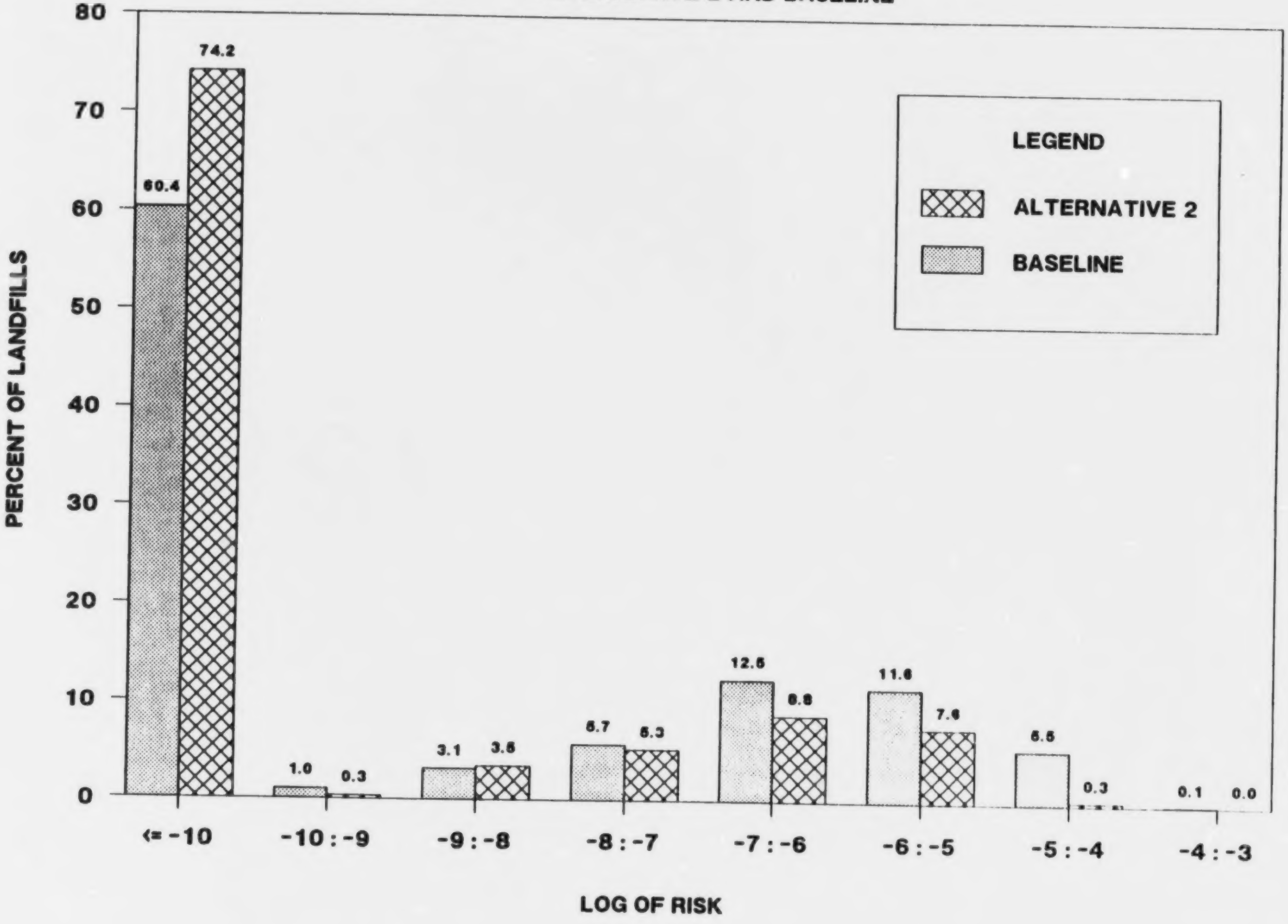
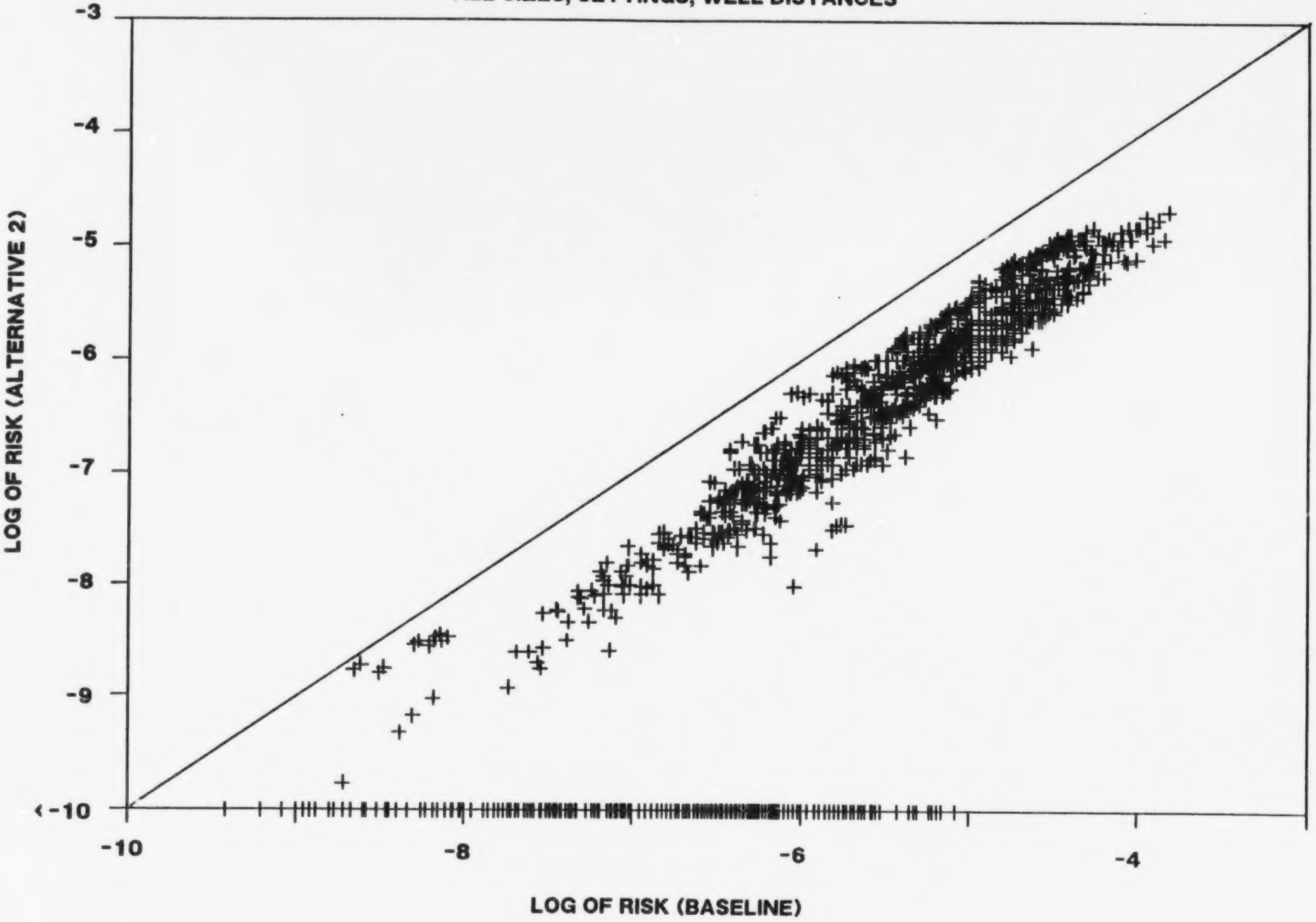


FIGURE IX-15

BASELINE VS. ALTERNATIVE 2 AVERAGE RISK

ALL SIZES, SETTINGS, WELL DISTANCES



The synthetic/composite liner, double leachate collection system, and composite cover reduce risk for several reasons. The cover reduces the amount of infiltration entering the landfill. Before leachate is released from the landfill, both synthetic membranes in the bottom liner must fail, which results in a lower LCS efficiency, and the leachate that is not collected must then travel through three feet of clay. As a result, releases are typically delayed for about 52 years in the 10-inch and 20-inch annual net infiltration regions, 74 years in the 1-inch region, and over 100 years in the 0.25-inch region. Because of this delay, some of the pollutant mass which would otherwise have been released is not released during the modeling period. The delay also results in additional pollutant degradation before release. Finally, the leachate collection systems remove some of the pollutant mass from the landfill.

This very stringent design does not reduce risk by a factor of more than about 20 in most scenarios. Most of the scenarios for which risk is eliminated are in the 0.25-inch net infiltration region, where the delay in release exceeds 100 years. Because we simulate risk resulting from only the first 100 years of release, these landfills have zero risk in our modeling. If we were to extend our pollutant release period, we would simulate positive (but small) risks for these landfills.

Population risks are 0.014 cancer cases per year (over the 300-year modeling period), down from a baseline of 0.077 cases per year.

Risk Reduction – Alternative 3

Under Alternative 3, landfills that are located in areas with greater than 40 inches of annual precipitation or that generate more than one foot of leachate during their active life are required to have leachate collection systems. All landfills must meet performance standards relating to time of travel to the uppermost aquifer and to infiltration of water through the cover. Landfill owners and/or operators can use a number of different liner/cover configurations, provided that they demonstrate the performance standards will be met.

To analyze the effectiveness of Alternative 3, we made some simplifying assumptions to assign each size/environmental setting/flow field scenario to one of four designs: unlined with vegetative cover, clay-lined with vegetative cover, unlined with synthetic cover, and synthetic-lined with synthetic cover. We assumed that there would be 30 years of post-closure care, including maintenance of the cover, leachate collection (for designs with an LCS), ground-water monitoring, and corrective action.

Figure IX-16 shows the distribution of 300-year average MEI risk for both the baseline and Alternative 3. The figure shows a shift in risk from the moderate and high risk ranges to the low and very low categories. Only 0.5 percent of the landfills have risks exceeding 10^{-5} under Alternative 3, compared with 5.6 percent in the baseline, and 9.6 percent have risks between 10^{-6} and 10^{-5}, compared with 11.6 percent in the baseline. The proportion of landfills with risks below 10^{-6} increases from about 83 percent in the baseline to about 90 percent under Alternative 3.

Figure IX-17 shows the risk reduction achieved under Alternative 3 for all combinations of landfill size, environmental setting, flow field, and well distance. The points plotted in this figure do not reflect the frequency of occurrence of each combination and do not include scenarios with baseline risk below 10^{-10}. As with the final rule and Alternative 1, risk remains unchanged from the baseline for many scenarios; points on the equal-risk line correspond to landfills that need only vegetative covers to comply with Alternative 3 design and performance standards. Unlike Alternative 1, the performance standards under Alternative 3 are not strictly risk-based, resulting in the need for liners and/or covers at some landfills with low baseline risk. We therefore see some risk reduction even at the low ends of the baseline risk range. Risk is generally reduced by a factor of 0 to 10 under Alternative 3, but the reduction is greater in some cases. According to our modeling, none of the landfills trigger corrective action under Alternative 3.

The expected number of cancer cases under Alternative 3 is 0.018 per year (over the 300-year modeling period), compared with 0.077 in the baseline.

As with the other regulatory alternatives, changing the duration of the post-closure care period from 30 to 10 or 40 years would increase or decrease simulated risk, but these changes could not be quantified.

Risk Reduction – Alternative 4

Alternative 4 requires ground-water monitoring and corrective action, but imposes no design standards. To analyze the effectiveness of this alternative, we assumed that corrective action would be triggered at all landfills for which constituent concentrations at the 10-meter well exceeded MCLs or, for pollutants without MCLs, the higher of the health-based level (10^{-5} risk-specific dose for carcinogens, reference dose for systemic pollutants) or the analytical detection limit. As with the other alternatives, the only corrective action technology that we simulate is ground-water recovery wells. We estimate that 35 percent of the landfills will trigger corrective action during the 30-year post-closure care period.

FIGURE IX-16
DISTRIBUTION OF AVERAGE RISK

ALTERNATIVE 3 AND BASELINE

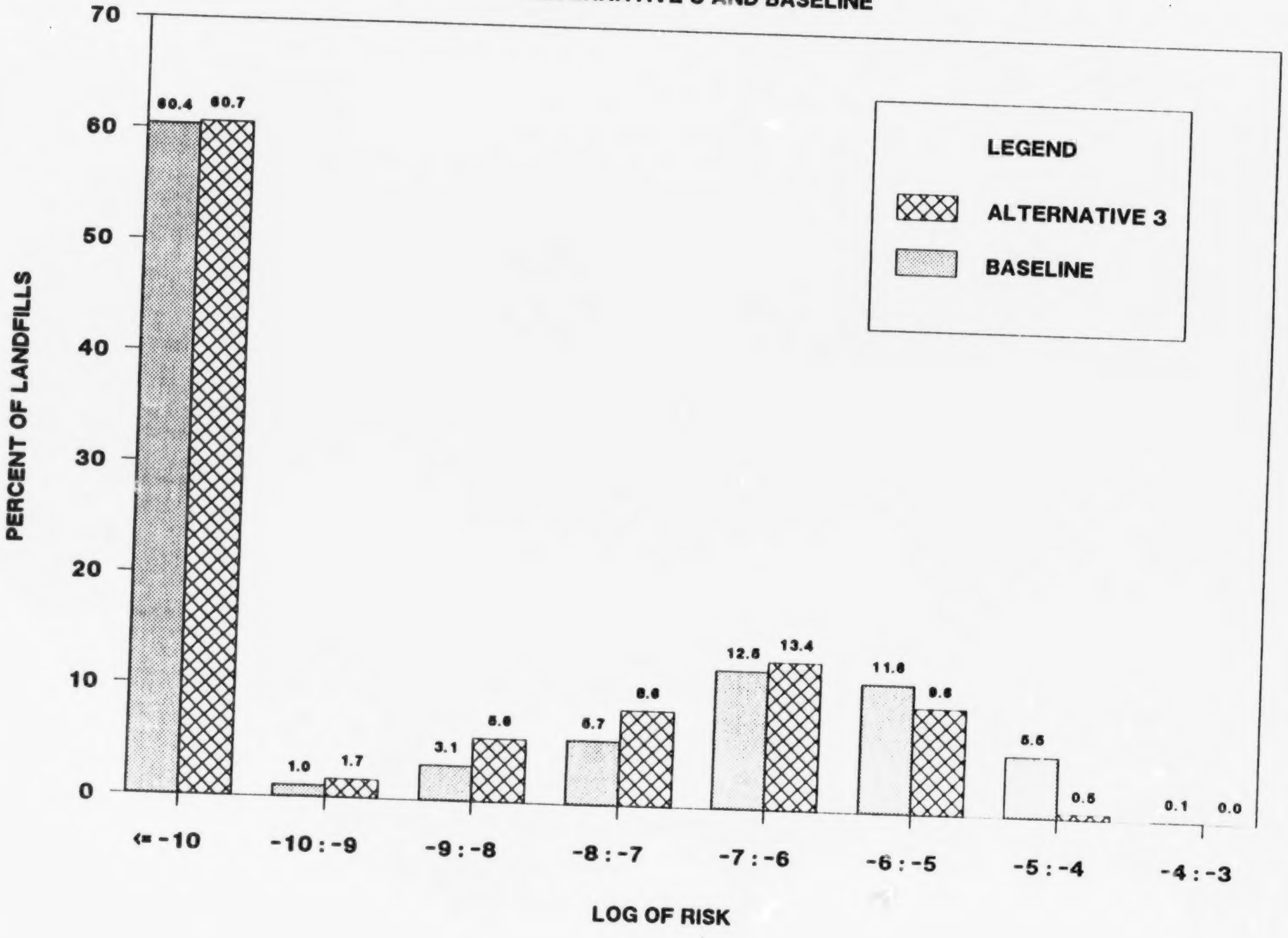


FIGURE IX-17

BASELINE VS. ALTERNATIVE 3 AVERAGE RISK

ALL SIZES, SETTINGS, WELL DISTANCES

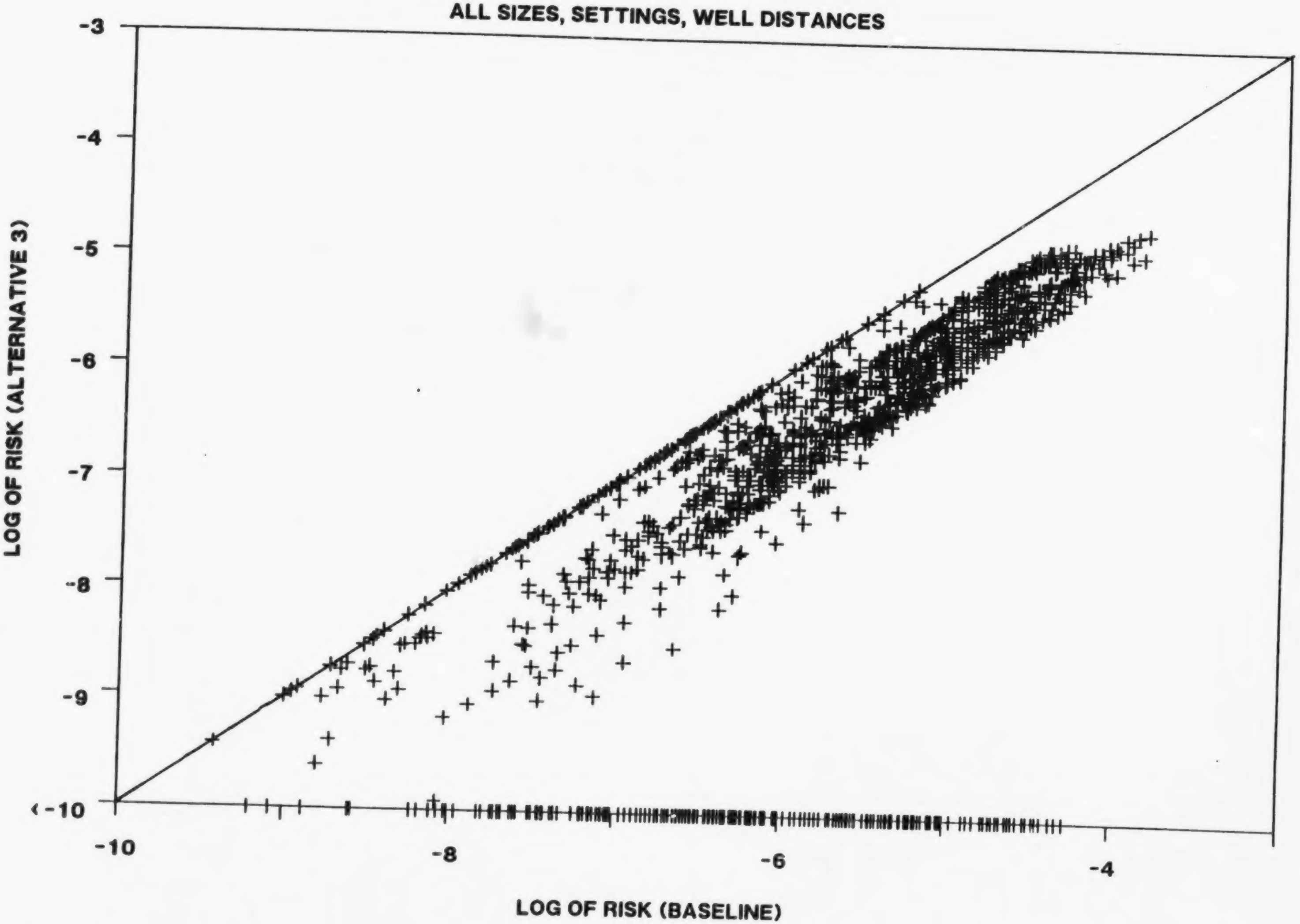


Figure IX-18 shows the distribution of 300-year average most exposed individual risk for both the baseline and Alternative 4. Under Alternative 4, the percentage of landfills with risks higher than 10^{-5} decreases from 5.6 to 2.1, and the percentage with risks between 10^{-6} and 10^{-5} decreases from 11.6 to 8.9 . This means that about 11 percent of the landfills have risks higher than 10^{-6} under Alternative 4, compared with 17 percent in the baseline.

Figure IX-19 shows the risk reduction achieved under Alternative 4 for all combinations of landfill size, environmental setting, flow field, and well distance. The points plotted in this figure do not reflect the frequency of occurrence of each combination and do not include scenarios with baseline risk below 10^{-10}. Risk remains unchanged for many scenarios; the points on the equal-risk line correspond to landfills that do not trigger corrective action (65 percent of all landfills). The magnitude of the risk reduction for landfills that trigger corrective action is highly variable, because the effectiveness of ground-water recovery wells varies significantly among flow fields and among well distances within a given flow field.

Population risks under Alternative 4 are 0.023 cancer cases per year (over the 300-year modeling period), down from a baseline of 0.077 cases per year.

According to our modeling, extending the duration of the post-closure care period from 30 to 40 years would have virtually no effect on simulated risk. Maintaining the integrity of the vegetative cover for an additional 10 years does not significantly reduce the amount of leachate generated over the modeling period. Moreover, all of the landfills that trigger corrective action with the unlined/vegetative cover design do so in the first 50 years (most, in fact, trigger in the first 30 years), so an additional 10 years of ground-water monitoring would not increase the number of landfills that perform corrective action. As discussed in Chapter VI, corrective action is generally triggered quickly in our modeling for unlined landfills because (1) the monitoring well is very close to the landfill, and (2) we make the conservative assumption that leachate releases occur in the first year of unit operation, rather than after the moisture storage capacity of the waste is filled.

Reducing the duration of the post-closure care period from 30 to 10 years would result in slightly higher human health risks than those presented above, because fewer landfills would trigger corrective action. However, the increase in risk would not be substantial, because the number of landfills that trigger corrective action would decrease by only 2 percent.

Risk Reduction – Summary

In this section, we compare the effectiveness of the final rule and the four alternatives in reducing risk to the maximum exposed individual (MEI) and to populations. As in the preceding sections, MEI risk refers to the maximum exposed individual lifetime risk averaged over the 300-year modeling period, and population risk is expressed as the number of cases expected annually over the modeling period.

FIGURE IX-18
DISTRIBUTION OF AVERAGE RISK

ALTERNATIVE 4 AND BASELINE

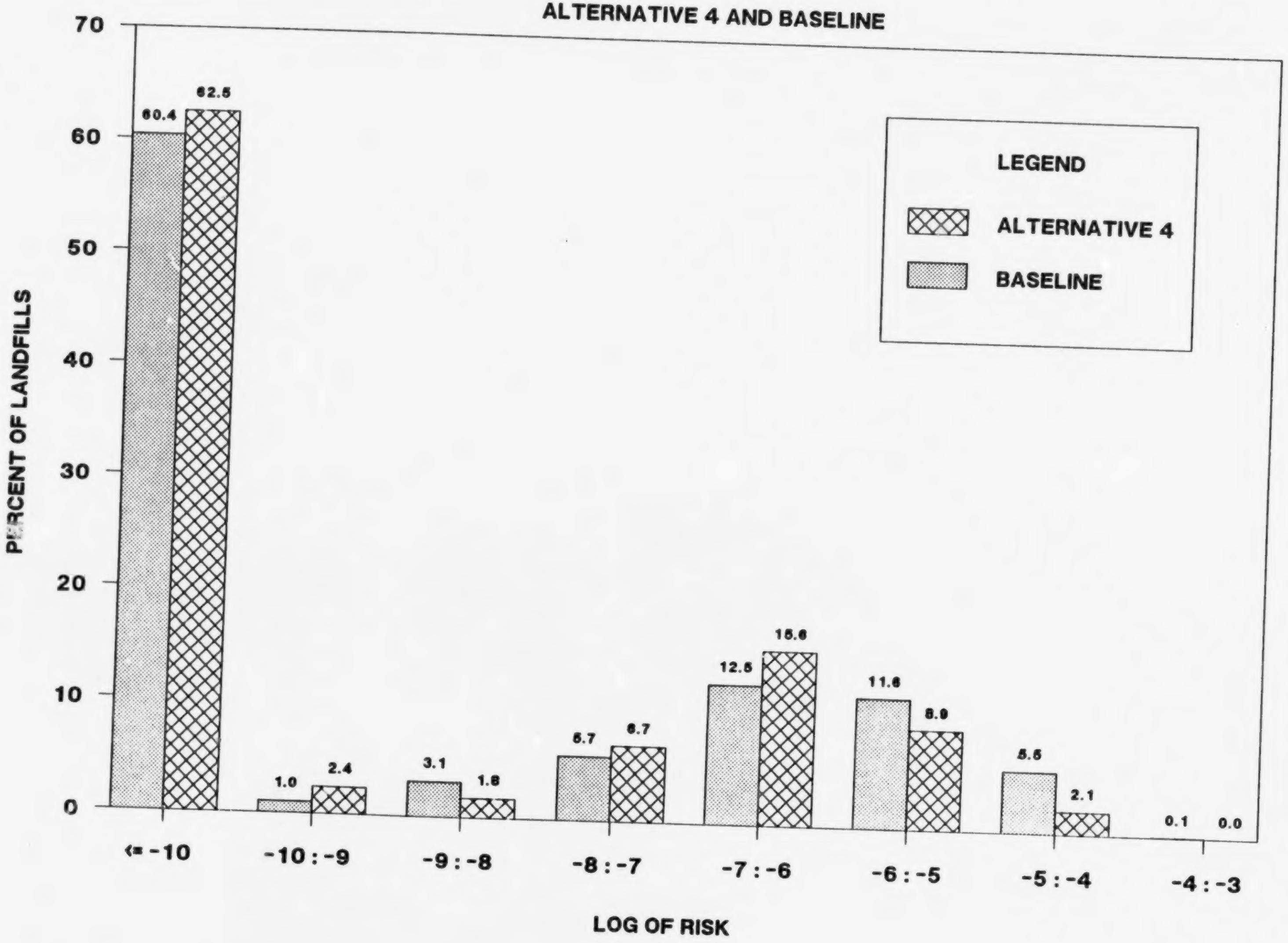


FIGURE IX-19

BASELINE VS. ALTERNATIVE 4 AVERAGE RISK

ALL SIZES, SETTINGS, WELL DISTANCES

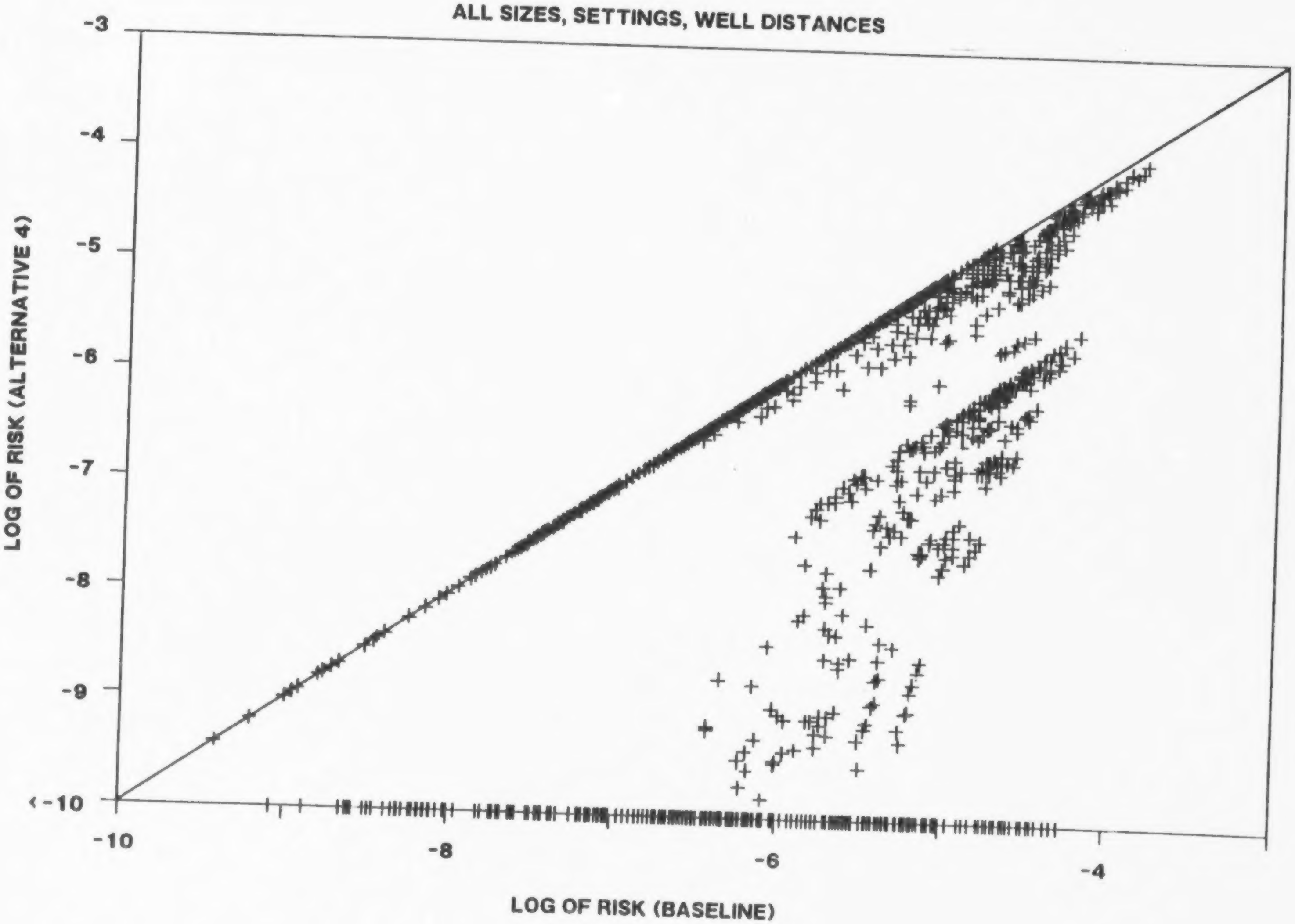


Figure IX-20 shows the cumulative frequency of average individual risk for the baseline, the final rule, and each alternative. The y-coordinate for any point on the graph corresponds to the percentage of landfills with risks exceeding the point's x-coordinate. We have excluded risks lower than 10^{-8} from the graph to provide greater resolution in the higher risk ranges.

The final rule and Alternatives 1 to 3 are effective in reducing the occurrence of high risks; all four of these options reduce the occurrence of risks higher than 10^{-5} from 5.6 percent of landfills in the baseline to less than 1 percent. Alternative 4 is less effective at reducing high risks since under this alternative about 2.1 percent of the landfills continue to have high risks. Alternative 4 is the only regulatory option that does not impose design or performance standards, relying only on response to releases (i.e., corrective action). The uncertainty of corrective action timing, efficiency, and cost may not be well reflected in these estimates.

For risks less than 10^{-5}, the rule and the alternatives have different effects on risk. Figure IX-20 shows that Alternatives 2 and 3 stay consistently below the other options: fewer landfills trigger risks at any given level under these alternatives. The final rule and Alternative 1 are very effective at reducing high risks, but for risks below 10^{-6}, they are similar to the baseline. Alternative 4 is slightly more effective at the low risk levels, but much less effective for risks greater than 10^{-5}.

FIGURE IX-20

CUMULATIVE FREQUENCY OF AVERAGE RISK

BASELINE, FINAL RULE, AND ALTERNATIVES

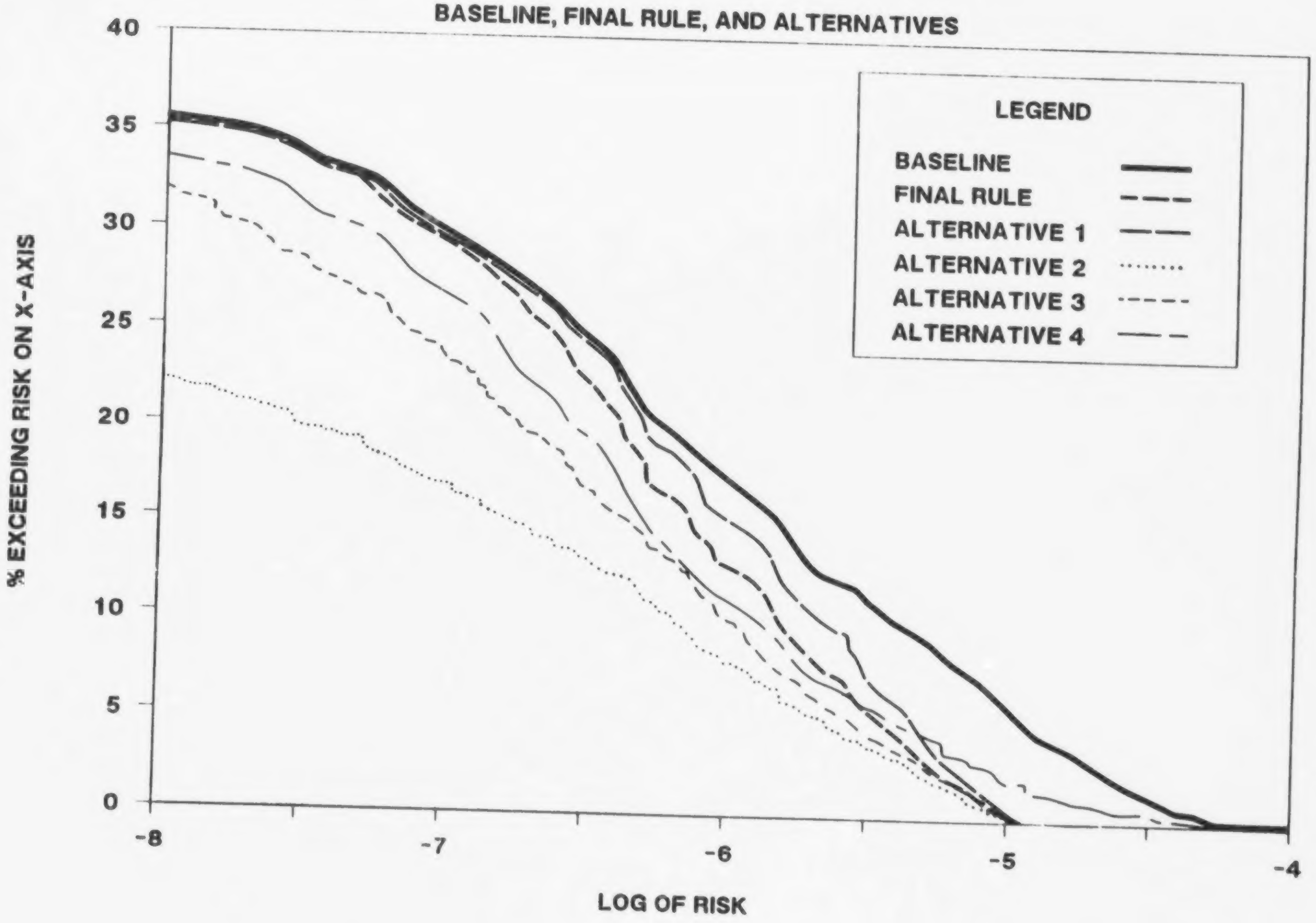


Table IX-2 shows the estimated number of cases expected annually over the 300-year modeling period, and the reduction in annual population risk for each option. Alternative 2 is the most effective of the options, reducing the number of cases by about 82 percent. Alternative 3 is slightly less effective, reducing population risks by about 77 percent. Alternative 4 and the final rule are about equally effective in reducing population risks; reductions are about 70 and 68 percent, respectively. Alternative 1 is the least effective regulatory option. The options are much more similar in their effectiveness in reducing population risks than they are in cost. The reduction in cases varies from the least effective to the most effective by a factor of only 1.28. At the same time, the cost of the most expensive option is approximately 4 times higher than that of the least expensive. We discuss the cost-effectiveness of each option in Chapter X.

Alternatives 2 and 3 result in the lowest population risks because they impose more stringent designs on all or most of the landfills. Under the final rule and Alternatives 1 and 4, many landfills can comply without liners and only vegetative covers. Nevertheless, all of the regulatory options are effective in reducing the occurrence of MEI risks greater than 10^{-5}.

Table IX-2

Predicted Population Risk Across 6,034 New Facilities

Regulatory Scenario	Cases per Year[1]	Reduction Cases per Year (percent)
Baseline	0.0770	--
Final Rule	0.0245	0.0525 (68.2%)
Alternative 1	0.0279	0.0491 (63.8%)
Alternative 2	0.0142	0.0628 (81.6%)
Alternative 3	0.0179	0.0591 (76.8%)
Alternative 4	0.0234	0.0536 (69.7%)

---

[1] Total population risk over the 300-year simulation period divided by 300.

If we took state regulatory requirements into account in the baseline, the number of cancer cases per year in the baseline would decline. Similarly, the number of cases under each option would decline. The number of cases *avoided* by each regulatory option would therefore be lower (i.e., the incremental benefit from the federal options would be less).

Resource Damage

In this section we present the results of the resource damage analysis for the baseline, the final rule, and each alternative. As discussed in Chapter IV, resource damage is measured as the cost to provide water to users whose supply is contaminated by releases from landfills. For this analysis, resource damage (i.e., replacement cost) is expressed in present-value terms.

The Subtitle D Risk Model evaluates damage in two ways. First, it calculates use value (i.e., the replacement costs for ground water currently used as a drinking water supply). Use value applies to all landfills reported to have ground-water wells within one mile. Second, it computes option value, the value of ground water that does not currently serve as a source of drinking water but that may be used in the future. Option value takes use value and multiplies it by the probability that, in a given future year, the aquifer will be used as a drinking water source. Because the probability is very low initially, and increases with time, option value is always lower than use value. Option value applies to landfills that currently do not have any downgradient wells within one mile as reported in the MSWLF survey.

Another important point concerning resource damage results is the impact of time and discounting. In our analysis of risk, a given value is the same whether it occurs in the first year of the simulation or the last year. When considering the value of a resource, though, we discount future events using discounted cash flow analysis. Two resource damage estimates, each occurring at a different time, may have the same undiscounted value. When discounted, however, the damage that occurs later will be worth less in present value terms. As a result of this discounting, the timing of plume formation has a significant impact on the resource damage results.

As discussed in Chapter VI, all of the Subtitle D Risk Model resource damage runs were performed with 30 years of post-closure care. In this chapter, we report resource damage results for post-closure care periods ranging from 10 to 40 years. To estimate resource damage under post-closure care periods different from those simulated in the model, we made two adjustments: (1) for unlined landfills, we adjusted the period of time during which corrective action could be triggered; (2) for lined landfills (which do not trigger corrective action in our modeling), we assumed that shortening the post-closure care period to 10 years would cause releases to begin 20 years earlier than those simulated in the model (and, conversely, that increasing the post-closure care period to 40 years would delay the onset of release by an additional 10 years), and we derived new resource damage results by adjusting the discounting of the water supply replacement costs.

Baseline Resource Damage

Our discussion of baseline resource damage parallels the discussion of baseline risk. We first examine the overall weighted distribution of resource damage, and then discuss the factors affecting baseline resource damage.

Distribution of Resource Damage

Figure IX-21 presents the distribution of resource damage for all units, expressed in present value terms. The numbers on the x-axis represent the upper bound of the resource damage interval (i.e., the bar labeled 0.4 includes landfills with replacement costs that are higher than $0.2 million but less than or equal to $0.4 million). These results have been weighted by all the relevant factors (e.g., facility size, infiltration rates). It shows that resource damage ranges from $0 to more than $4 million. The majority of landfills, though, have resource damages valued at less than $200,000. The model predicts that about 71 percent of landfills have resource damage. Only 31 percent of landfills have resource damage exceeding $200,000, and about 13 percent have resource damage in excess of $1 million. The figure also shows how the distribution of resource damage divides into use value and option value. Because option value is based on the probability that a ground-water source may someday be used, it tends to be much lower than use value for a given set of conditions. In fact, the model data indicate that option value is, on average, only a tenth of use value. The result is that option value dominates at lower levels of resource damage while use value is the only measure to appear at levels exceeding $400,000.

Figure IX-22 summarizes the resource damage estimates in terms of a cumulative frequency distribution. This graph includes both use and option value. The median resource damage is about $76,500; 13 percent of the landfills have damages exceeding $1 million, and seven percent exceed $2 million.

Figure IX-23 presents the cumulative frequency distribution for use value alone (i.e., landfills with wells within one mile). The median replacement cost for this subset of landfills is about $485,000, and about 28 percent of these landfills have damages that exceed $1 million.

The total resource damage for both use and option value for 6,034 new landfills is about $2.58 billion.

FIGURE IX-21

DISTRIBUTION OF RESOURCE DAMAGE

BASELINE

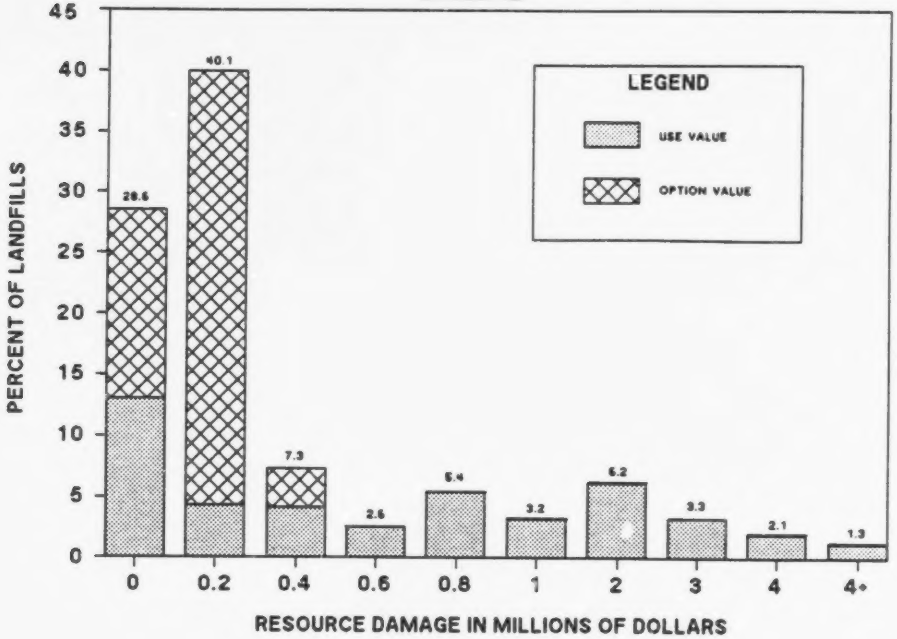


FIGURE IX-22

CUMULATIVE FREQUENCY OF RESOURCE DAMAGE

BASELINE: USE VALUE AND OPTION VALUE

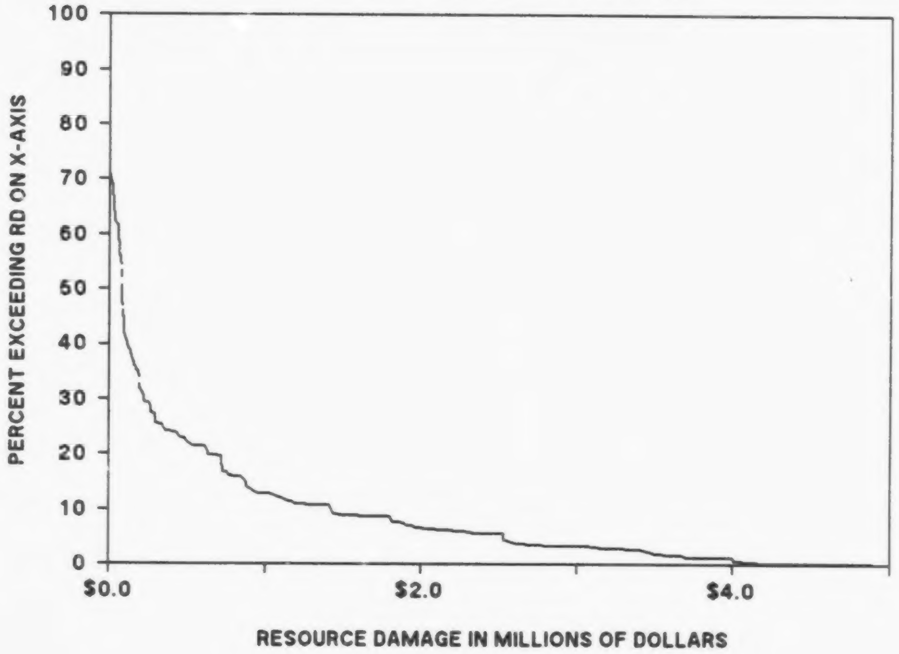
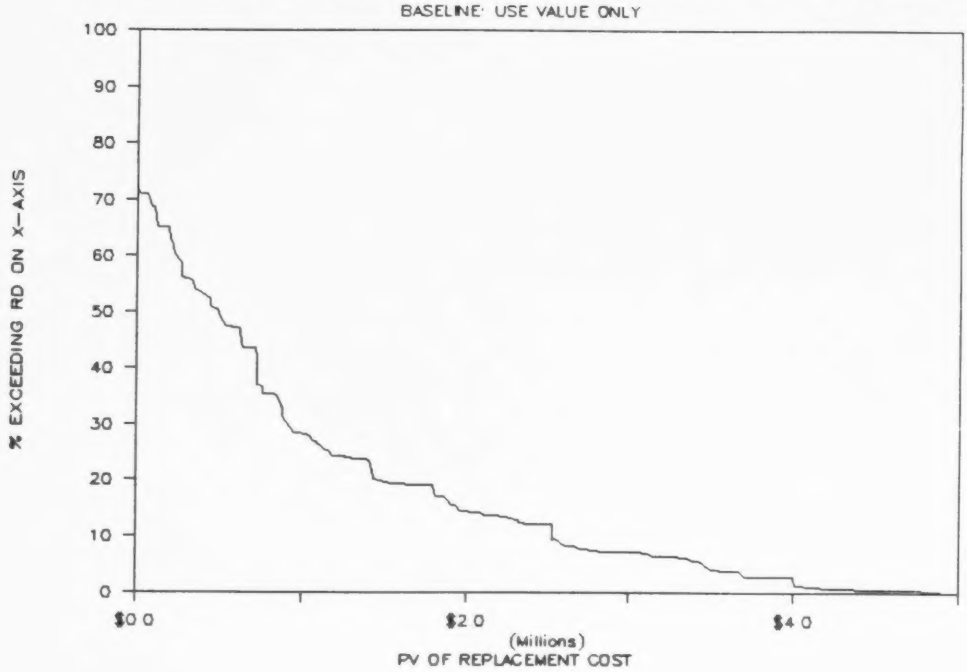


FIGURE IX-23

CUMULATIVE FREQUENCY OF RESOURCE DAMAGE



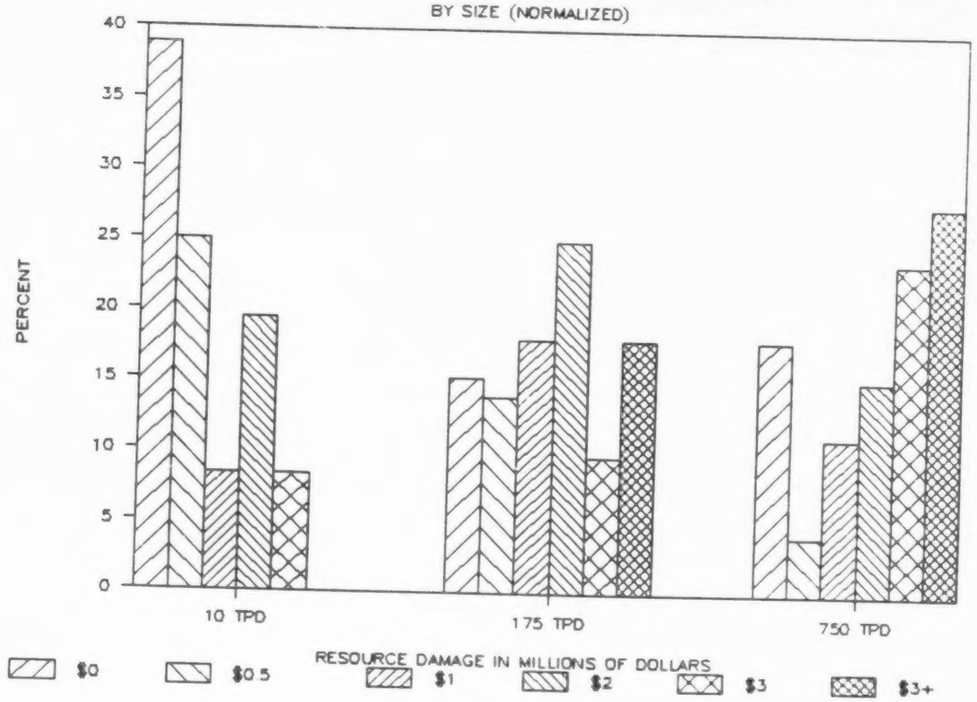
Factors Affecting Resource Damage

The Subtitle D model estimates resource damage for landfills of different sizes in a variety of environmental and hydrogeological settings. Because resource damage is affected primarily by plume size and timing we will assess the impact of size and environmental setting in terms of these two parameters. A preliminary analysis of use value and option value indicated that the two are highly correlated (Pearson's r = .73). Consequently, we decided that analysis of only use value would be sufficient to illustrate how resource damage varies among specific environmental factors. Thus, the percentages refer only to landfills that have wells within 1,600 meters.

Facility Size. Figure IX-24 shows that landfill size has a large impact on resource damage. Less than 30 percent of the 10-TPD scenarios have resource damage greater than $1 million. The plurality of scenarios with small landfills have no resource damage. In sharp contrast, 66 percent of the 750-TPD scenarios have resource damage greater than $1 million and nearly 30 percent of the scenarios have damage in excess of $3 million. Less than 20 percent have no resource damage. The greater mass of waste and larger area of the large facilities leads to larger plumes and, as a direct result, higher resource damage. The high resource damage profile of the large landfills is mitigated by their small proportion of the landfill population.

FIGURE IX-24

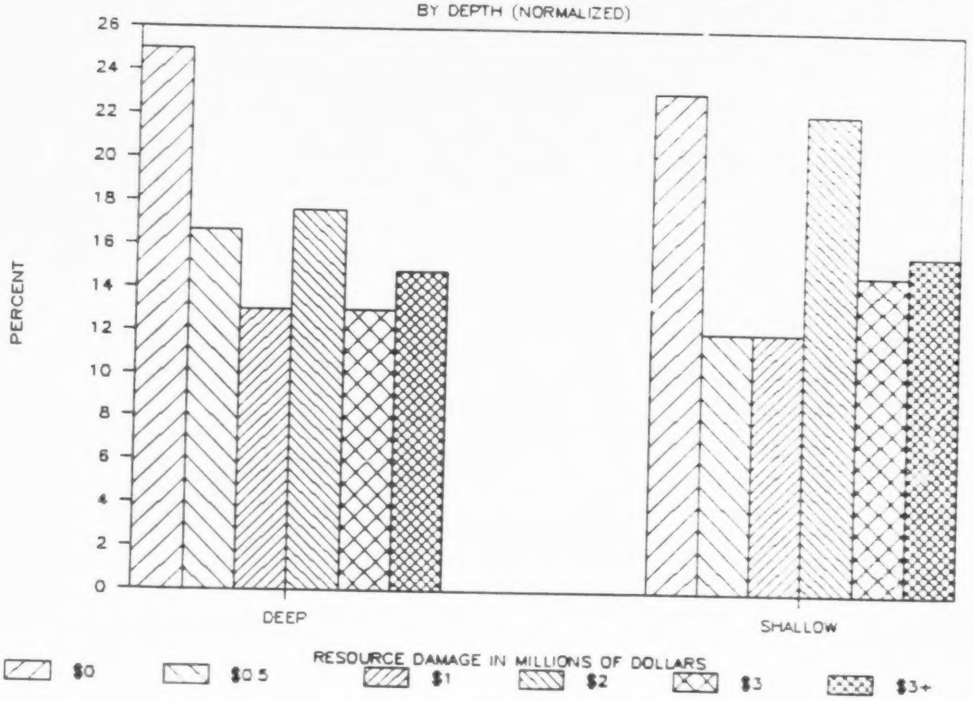
DISTRIBUTION OF RD - BASELINE



Water-Table Depth. Water table has only a small impact upon resource damage. The slight differences in the resource damage profiles for shallow and deep water tables (Figure IX-25) are probably a function of discounting. In shallow water tables, leachate breaks through the unsaturated zone more quickly, resulting in earlier formation of the plume. Consequently, resource damages are slightly higher for the shallow setting.

FIGURE IX-25

DISTRIBUTION OF RD - BASELINE



Net Infiltration. Net infiltration has a strong effect on source damage. Infiltration affects both plume size and the timing of plume development. In wetter climates plumes tend to be larger and to occur sooner because greater pollutant mass is released earlier in the modeling period.

Figure IX-26 shows the evidence of this effect. In the driest setting, 0.25-inch, most of the scenarios have no resource damage, and the number steadily decreases in the higher intervals. This pattern reverses in the 20-inch setting where a growing proportion of scenarios falls into the higher resource damage intervals. When we weight the four profiles by the proportion of landfills located in each infiltration region, however, the 0.25- inch and 10-inch settings have the most effect, mitigating the impact of the 20-inch scenarios.

Flow Fields. Aquifer characteristics also affect resource damage. In the lowest velocity aquifers, plumes grow relatively slowly, but pollutant concentrations remain relatively high. In the highest velocity aquifers, plumes grow rapidly but also dissipate rapidly. In the moderate velocity flow fields, plumes grow rapidly and remain above threshold concentrations for a long period. Consequently, lower resource damages occur at either extreme, and higher resource damages occur in the middle velocities, as shown by Figure IX-27. Flow fields A, D, and E, the slowest and the two fastest aquifers, respectively, have lower resource damage profiles. On the other hand, the moderate-velocity flow fields B and C have the highest resource damage profiles.

The double-layer flow fields follow a similar pattern. Flow fields G and H behave like B and C except that the top aquifer delays plume development and dilutes contaminant concentrations (plume size is estimated for the lower layer). Figure IX-27 shows a slight shift to low resource damage resulting from these effects. Flow fields F and K (confined aquifers) behave like hybrids between A and C and between A and B. The aquitards delay and disperse contamination, producing lower resource damage profiles. Flow fields A, B, C, and G have the highest proportion of landfills. Consequently, they have the greatest effect on weighted, overall resource damage.

Summary. Estimates of resource damage are heavily dependent on the current status of ground-water use, plume size, and the timing of contamination. Because ground water in the vicinity of more than half the landfills is not currently used, most contamination causes resource damage that has relatively low present value. In some cases, however, resource damage can be extensive, valued at as much as $5 million. Environmental factors have an impact on resource damage by affecting plume size and its timing.

FIGURE IX-26

DISTRIBUTION OF RD - BASELINE

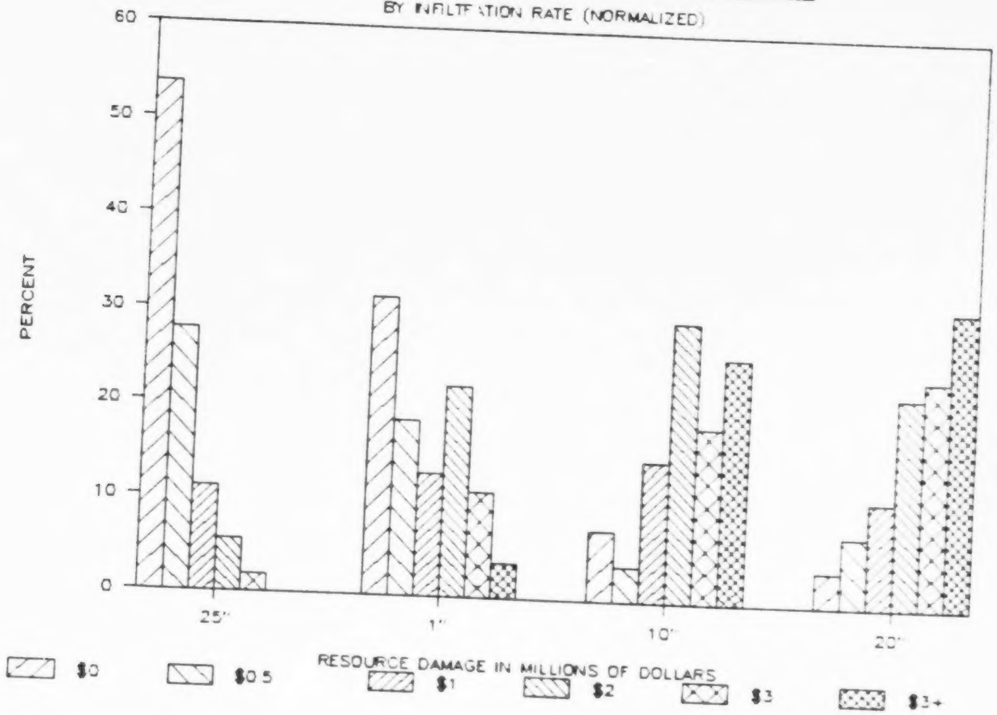
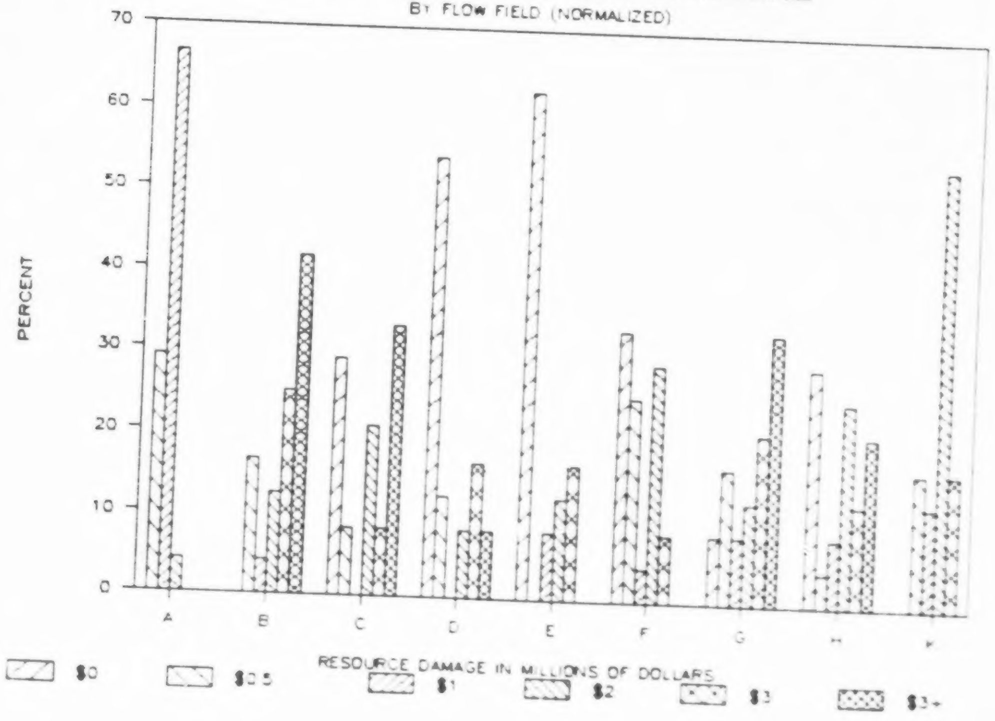


FIGURE IX-27

DISTRIBUTION OF RD - BASELINE



Resource Damage Reduction – Final Rule

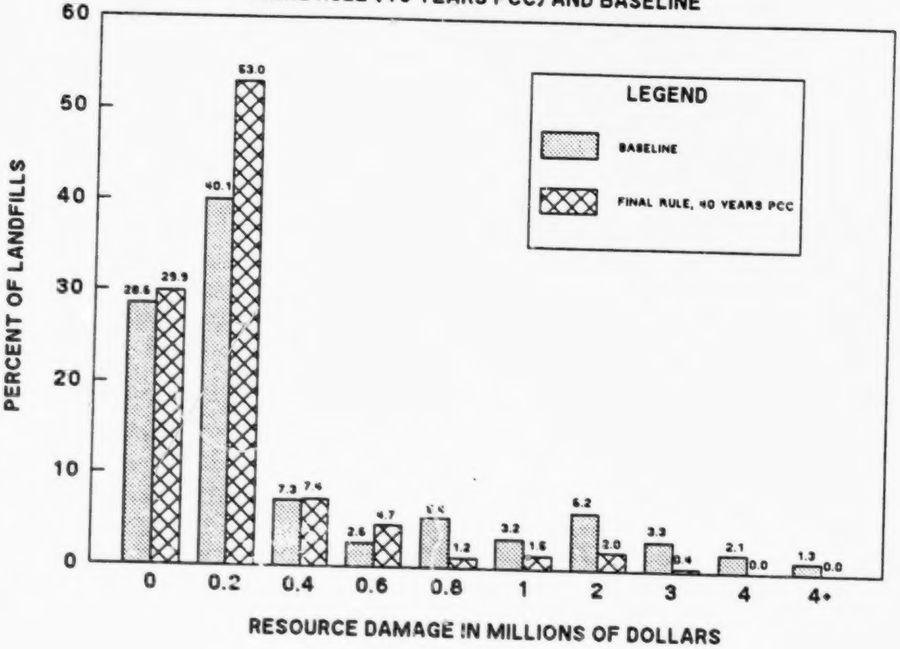
Under the final rule, landfills must comply with a uniform design standard or MCL-based performance standard. Under our modeling assumptions, we assigned 56 percent of landfills no liner and only a vegetative cover; less than 1 percent were assigned synthetic covers at closure; and 43 percent were assigned composite or synthetic liners and synthetic covers.

Figure IX-28 shows the distribution of resource damage for the baseline and the final rule; the first graph assumes that the post-closure care period is 40 years long, while post-closure care is assumed to last for only 10 years in the second graph. In both cases, the resource damage distributions shift to the lower replacement cost intervals under the final rule. In the baseline, about 13 percent of the landfills have resource damages exceeding $1 million (present value). If the post-closure care period is 40 years long, only 2.5 percent of landfills have resource damages above $1 million under the final rule, while nearly 6 percent of landfills do so if the post-closure care period is reduced to 10 years. The percentage of landfills with resource damage costs at or below $200,000 is about 69 percent in the baseline, compared to 72 to 83 percent under the final rule (10-year and 40-year post-closure care periods, respectively).

The median water supply replacement cost under the final rule ranges from $31,500 (40-year post-closure care period) to $38,000 (10-year post-closure care period), compared to $76,500 in the baseline. The total resource damage across all 6,034 landfills under the final rule is $0.84 billion and $1.24 billion for the 40- and 10-year post-closure care periods, respectively; this represents a $1.74 billion to $1.34 billion reduction from the baseline total of $2.58 billion.

FIGURE IX-28
DISTRIBUTION OF RESOURCE DAMAGE

FINAL RULE (40 YEARS PCC) AND BASELINE



FINAL RULE (10 YEARS PCC) AND BASELINE

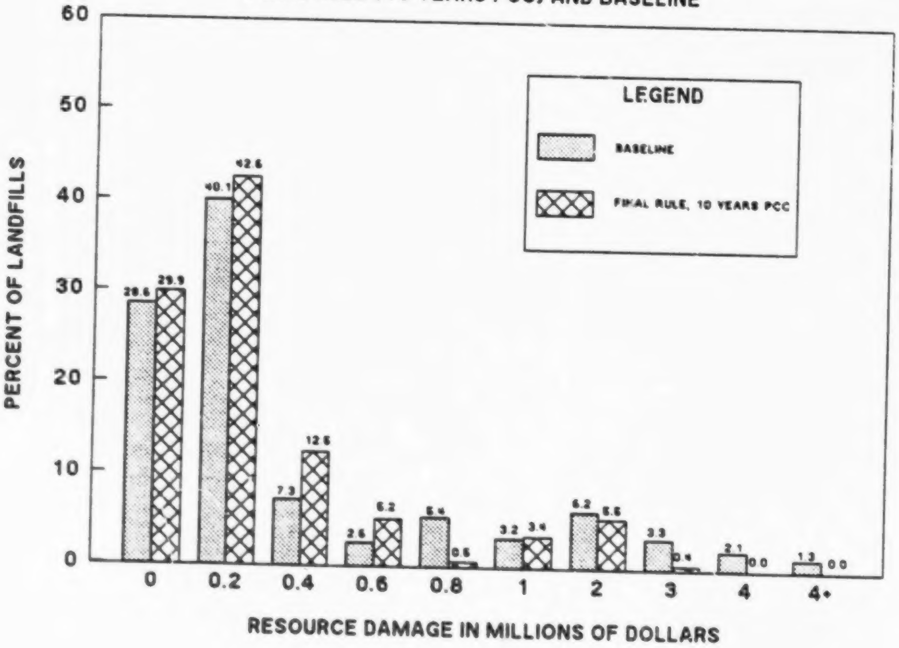


Figure IX-29 shows the reduction in resource damage achieved under the final rule for all combinations of landfill size, environmental setting, and flow field, and for two measures of resource damage: use value and option value. The points plotted on these graphs do not reflect the frequency of occurrence of each combination. The graphs show the effects of applying different designs and post-closure care periods to landfills. Landfills that meet the performance standard in the baseline (i.e., with vegetative covers and no liners) have no reduction in resource damage (i.e., points on the 45-degree line). The greatest reductions occur at landfills constructed with both synthetic liners and synthetic covers. We assume that these containment systems typically remain intact during the 20-year active life and 10- to 40-year post-closure care period. All other factors being equal, the 30-year delay in leachate releases provided under the 10-year post-closure care period reduces the present value of the replacement costs by 59 percent; the 60-year delay provided under the 40-year post-closure care period reduces replacement costs by 83 percent. Moreover, the delay in release can affect the size of the plume by reducing the amount of pollutant mass released during the modeling period.

Resource Damage Reduction – Alternative 1

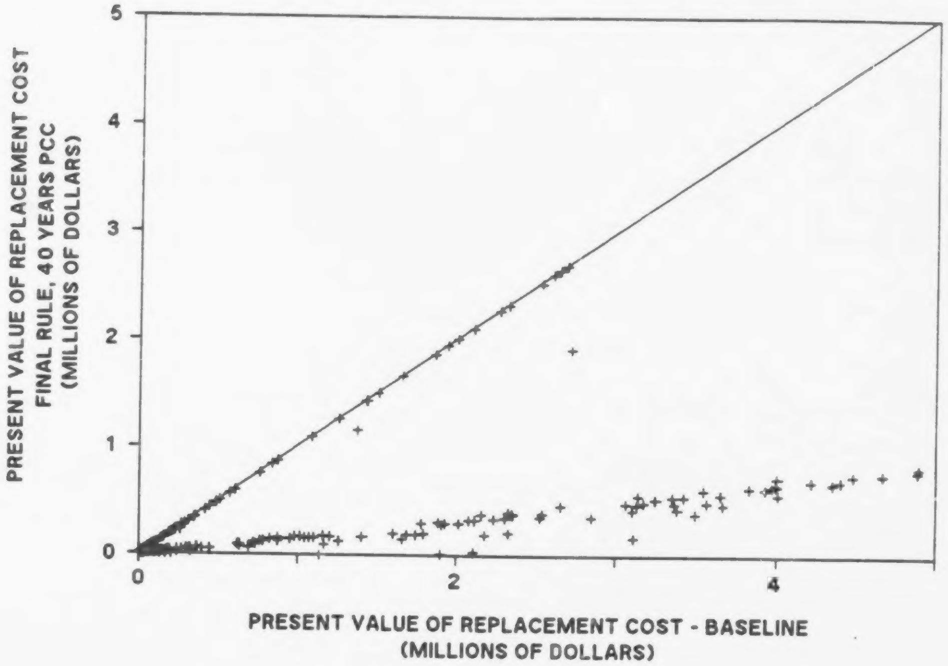
The modeled version of Alternative 1 assumes that landfills must comply with a 10^{-5} design goal at the unit boundary. As discussed previously, we estimate that vegetative covers alone are sufficient to meet the performance standard at 61 percent of all landfills; 11 percent require synthetic covers; and 28 percent need synthetic liners, LCSs, and synthetic covers. The corrective action trigger for Alternative 1 is set at the 10^{-5} risk level as well. This trigger is different from the trigger used for the final rule and Alternatives 2 through 4. In our modeling, none of the landfills trigger corrective action under this alternative.

Figure IX-30 shows the distribution of resource damage for both the baseline and Alternative 1. Under Alternative 1, none of the landfills have resource damage in excess of $3 million, compared to over 3 percent in the baseline. The percent of landfills with resource damage above $1 million is about 13 percent in the baseline, and 6.6 to 6.8 percent under Alternative 1 (40- and 30-year post-closure care periods, respectively). The percentage of landfills with resource damage costs of $200,000 or less increases from about 69 percent in the baseline to 78 to 79 percent under Alternative 1.

The median replacement cost under Alternative 1 ranges from $35,000 to $37,500 (depending on the duration of the post-closure care period), compared to $76,500 in the baseline. The total resource damage under Alternative 1 is $1.21 billion to $1.29 billion, representing a $1.37 billion to $1.29 billion reduction in resource damage from the baseline of $2.58 billion.

FIGURE IX-29
REDUCTION IN RESOURCE DAMAGE

FINAL RULE (40 YEARS PCC) COMPARED TO BASELINE



FINAL RULE (10 YEARS PCC) COMPARED TO BASELINE

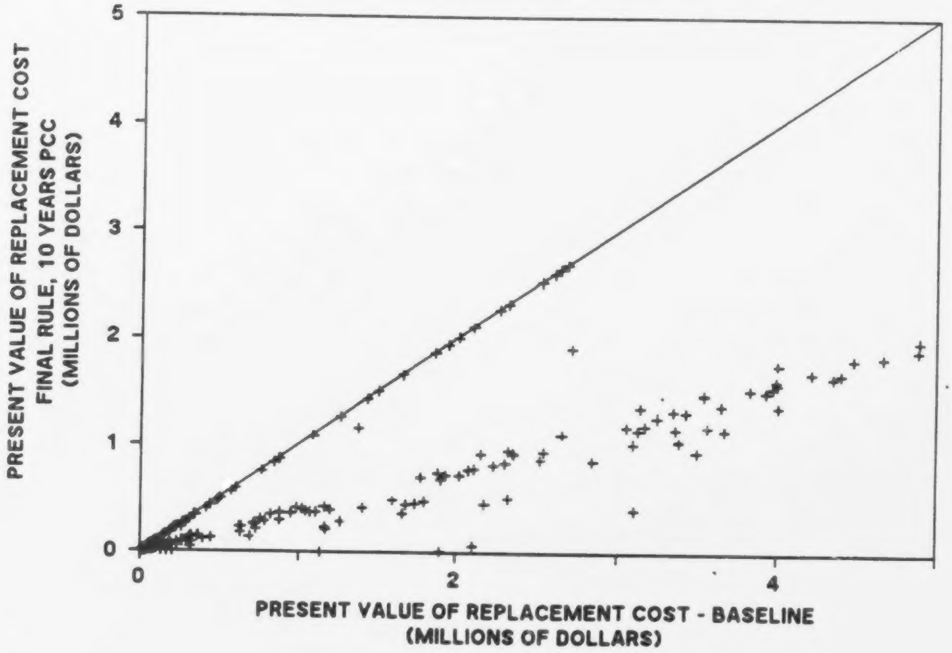
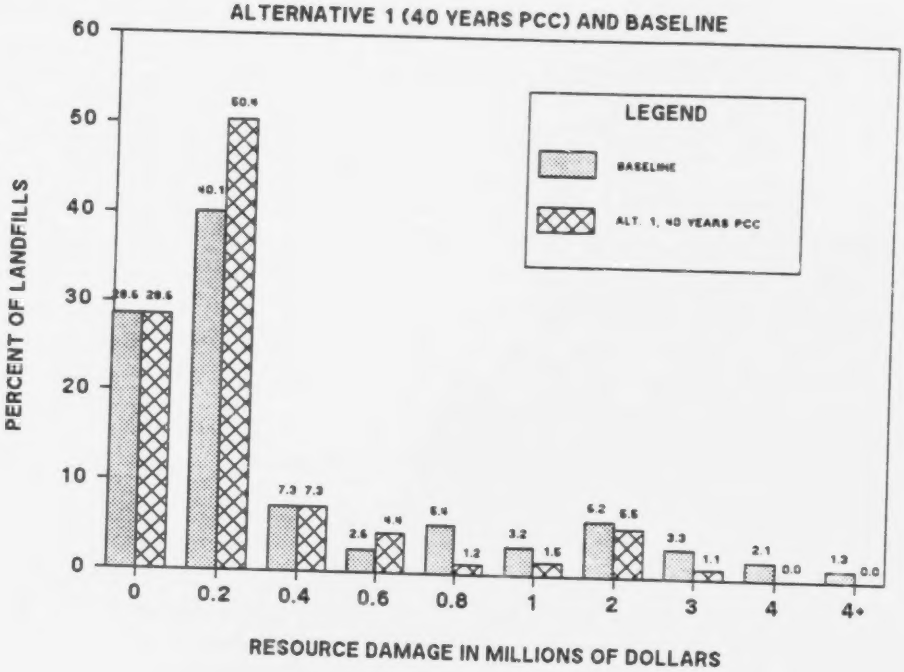


FIGURE IX-30

DISTRIBUTION OF RESOURCE DAMAGE


ALTERNATIVE 1 (40 YEARS PCC) AND BASELINE

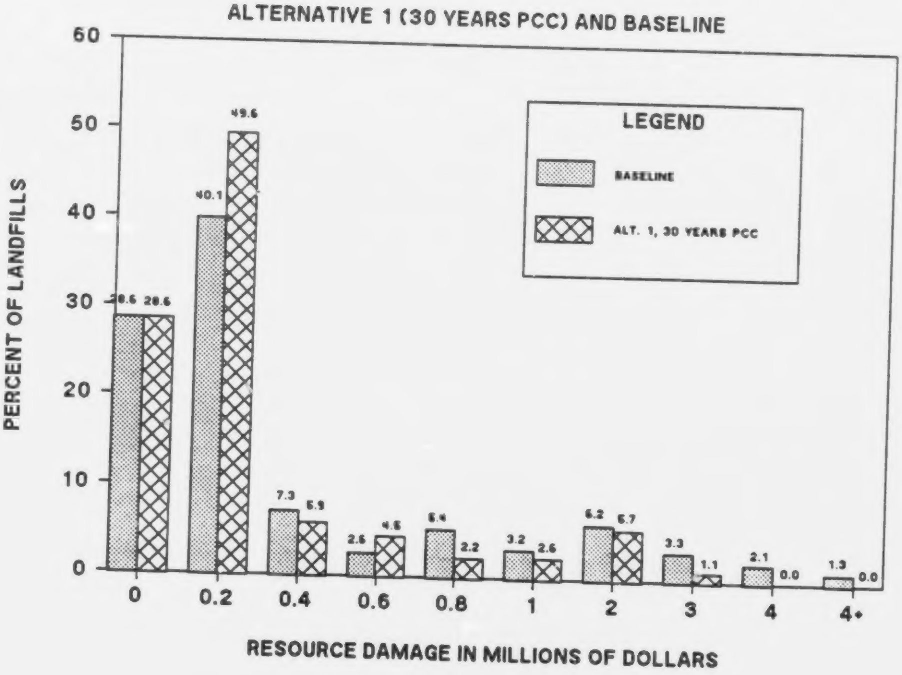

ALTERNATIVE 1 (30 YEARS PCC) AND BASELINE

Figure IX-31 shows the reduction in resource damage achieved under Alternative 1. As with the final rule, points on the equal-risk line represent landfills that can comply with the risk-based performance standard with no liners and only vegetative covers. A second group of points lying just below the 45-degree line represents unlined landfills with synthetic covers. A third group of points lying well below the 45-degree line represents landfills that have synthetic liners and synthetic covers; this design is quite effective in reducing resource damage because it delays the onset of releases. The reduction is slightly higher under the 40-year post-closure care period than the 30-year post-closure care period. If all other factors are held constant, increasing the post-closure care period from 30 to 40 years decreases the present value of the resource damage from all landfills by 26 percent.

Resource Damage Reduction – Alternative 2

Alternative 2 requires all new landfills to have a synthetic/composite liner, two leachate collection systems, and a composite cover. Corrective action is never triggered under this alternative in our modeling.

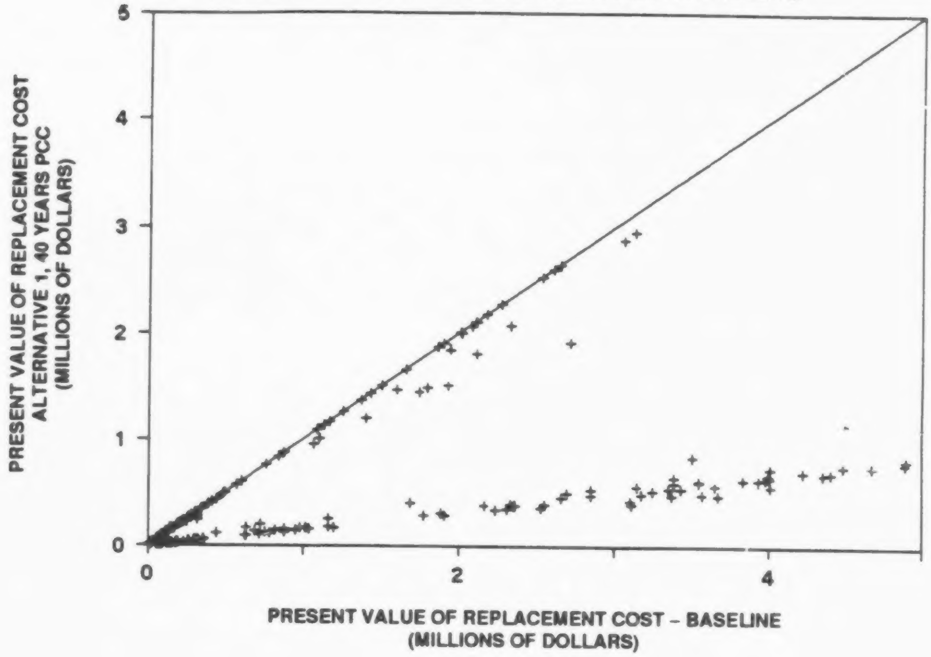
Figure IX-32 shows the distribution of resource damage both from the baseline and with Alternative 2 requirements in place. Over half of the landfills have zero resource damage under this alternative, regardless of the duration of the post-closure care period. The median water supply replacement cost is therefore zero. If the post-closure care period is 40 years long, no landfills have replacement costs exceeding $800,000, compared to over 16 percent in the baseline. If the post-closure care period is reduced to 10 years. resource damage under Alternative 2 can be as high as $2 million.

Figure IX-33 clearly shows that resource damage is reduced for every landfill regardless of size, environmental setting, or flow field. All points are significantly below the 45-degree line. Alternative 2 reduces resource damage more effectively under longer post-closure care periods. All other factors being equal, lengthening the post-closure care period from 10 to 40 years causes the present value of the resource damage from all landfills to decrease by 59 percent.

The total resource damage for 6,034 landfills under Alternative 2 ranges from $0.28 billion (assuming 40 years of post-closure care) to $0.69 billion (assuming 10 years of post-closure care). This represents a reduction of $2.3 billion to $1.89 billion from the baseline total of $2.58 billion.

FIGURE IX-31
REDUCTION IN RESOURCE DAMAGE

ALTERNATIVE 1 (40 YEARS PCC) COMPARED TO BASELINE



ALTERNATIVE 1 (30 YEARS PCC) COMPARED TO BASELINE

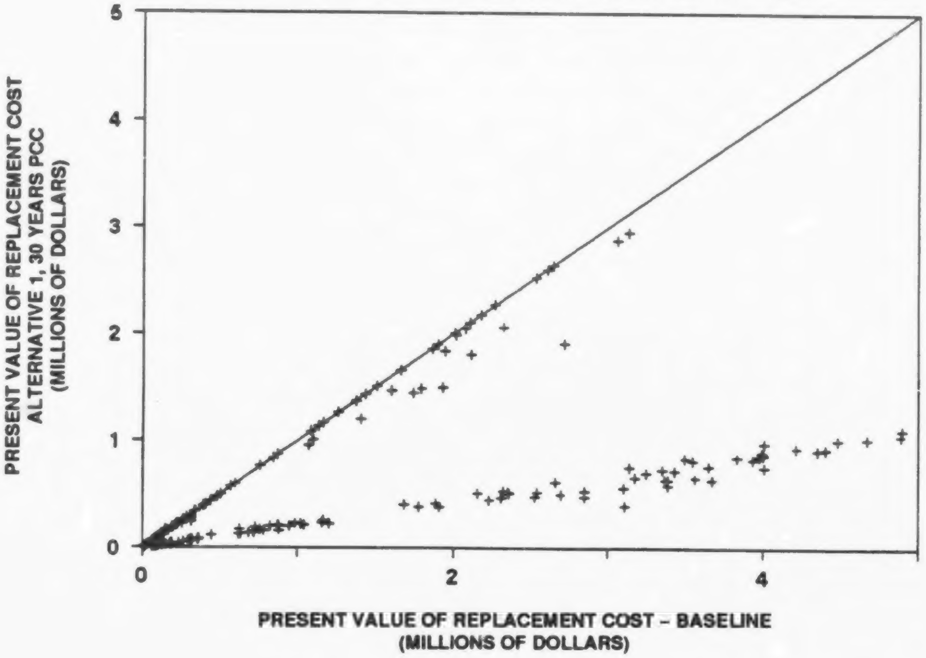
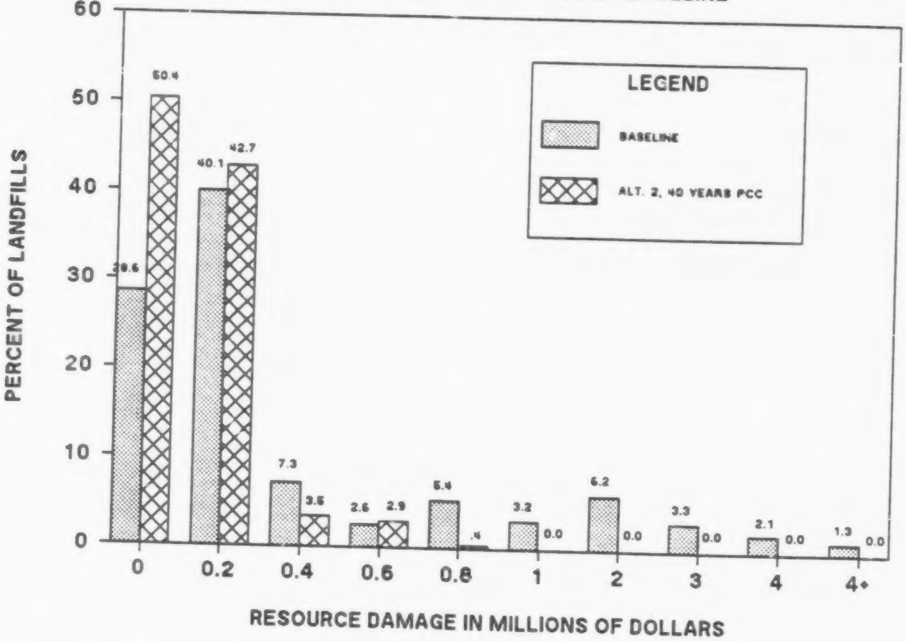


FIGURE IX-32
DISTRIBUTION OF RESOURCE DAMAGE

ALTERNATIVE 2 (40 YEARS PCC) AND BASELINE



ALTERNATIVE 2 (10 YEARS PCC) AND BASELINE

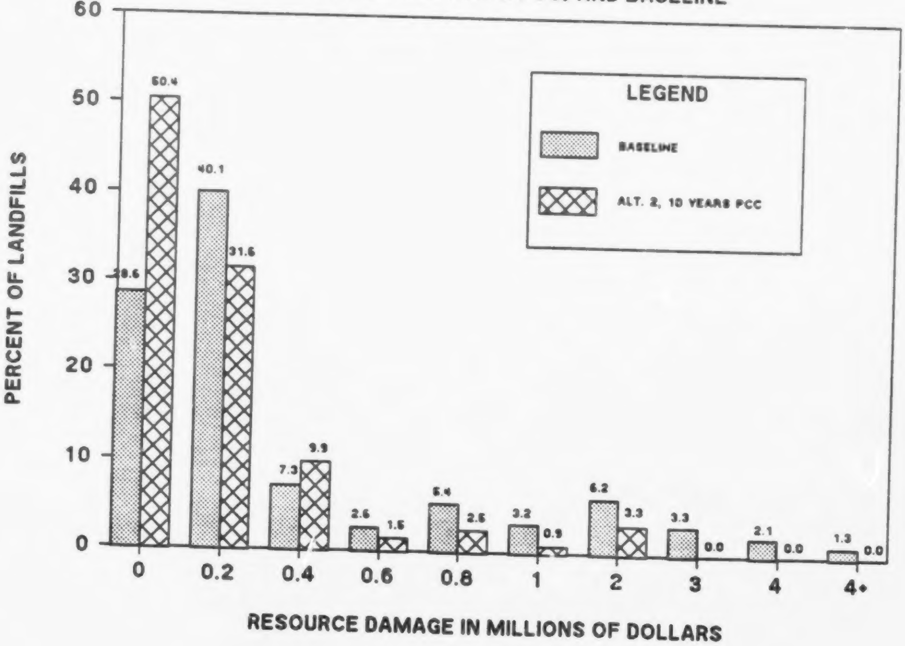
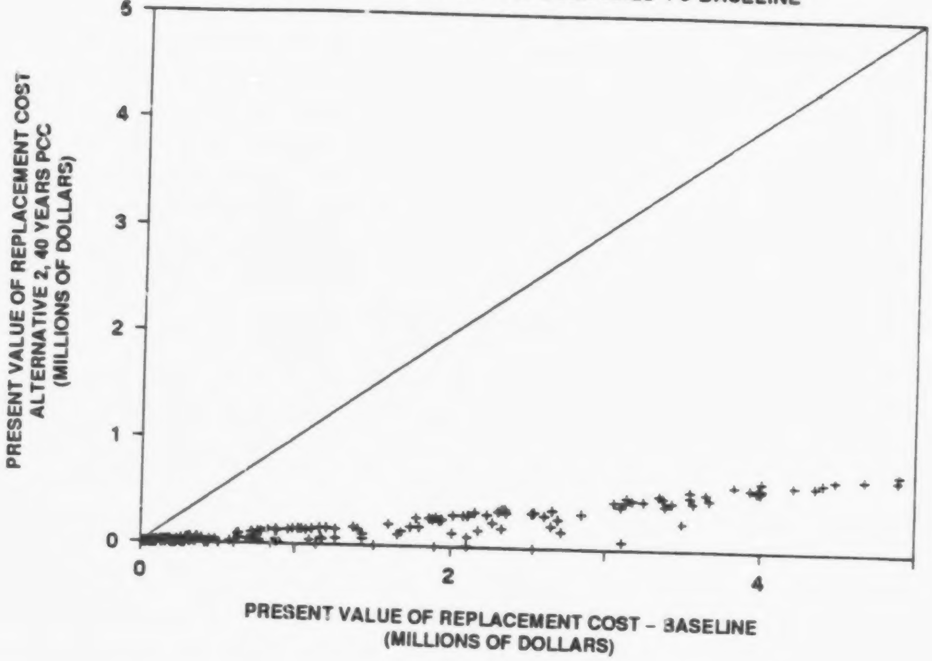
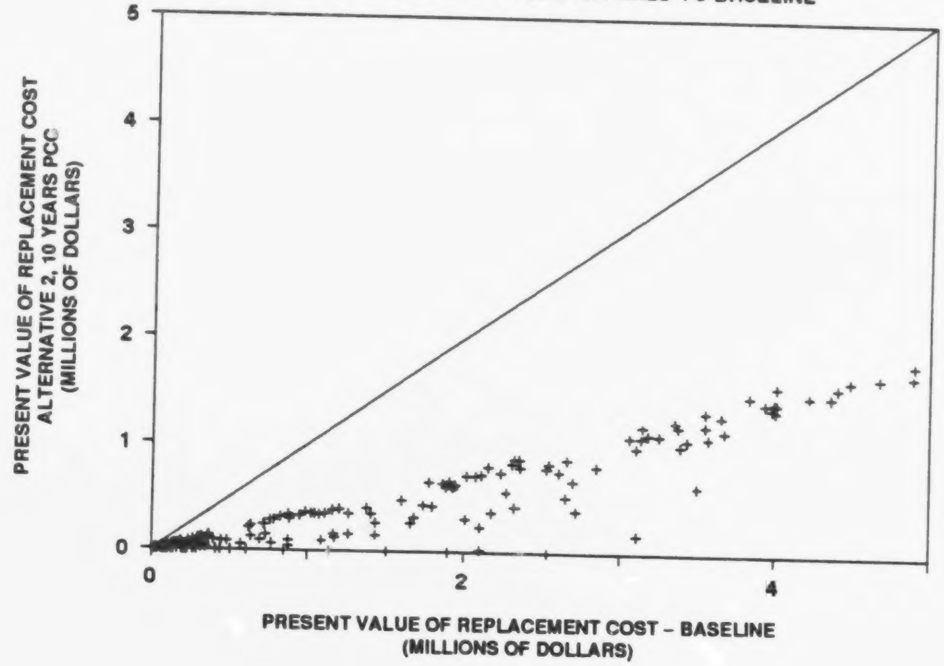


FIGURE IX-33
REDUCTION IN RESOURCE DAMAGE



ALTERNATIVE 2 (40 YEARS PCC) COMPARED TO BASELINE



ALTERNATIVE 2 (10 YEARS PCC) COMPARED TO BASELINE

Resource Damage Reduction – Alternative 3

Under Alternative 3, landfills must meet performance standards related to time of travel to the uppermost aquifer and infiltration of water through the cover. If the landfill generates more than a foot of leachate during its active life, it must have a leachate collection system. As discussed earlier, we made some simplifying assumptions to assign landfills to one of four designs: unlined/vegetative cover, clay-lined/vegetative cover, unlined/synthetic cover, and synthetic-lined/synthetic cover. According to our modeling, corrective action is never triggered under this alternative.

As shown in Figure IX-34, Alternative 3 is nearly as effective as Alternative 2. If the post-closure period is 40 years long, less than 1 percent of the landfills have resource damage exceeding $1 million, compared to about 13 percent in the baseline. About 90 percent of landfills have resource damage of $200,000 or less under Alternative 3 if post-closure care is conducted for 40 years, an increase from 69 percent in the baseline. If the post-closure care period is reduced to 10 years, the effectiveness of Alternative 3 is somewhat lower. Under this scenario, 3.5 percent of landfills have resource damage exceeding $1 million, and 79 percent of landfills have resource damage of $200,000 or less.

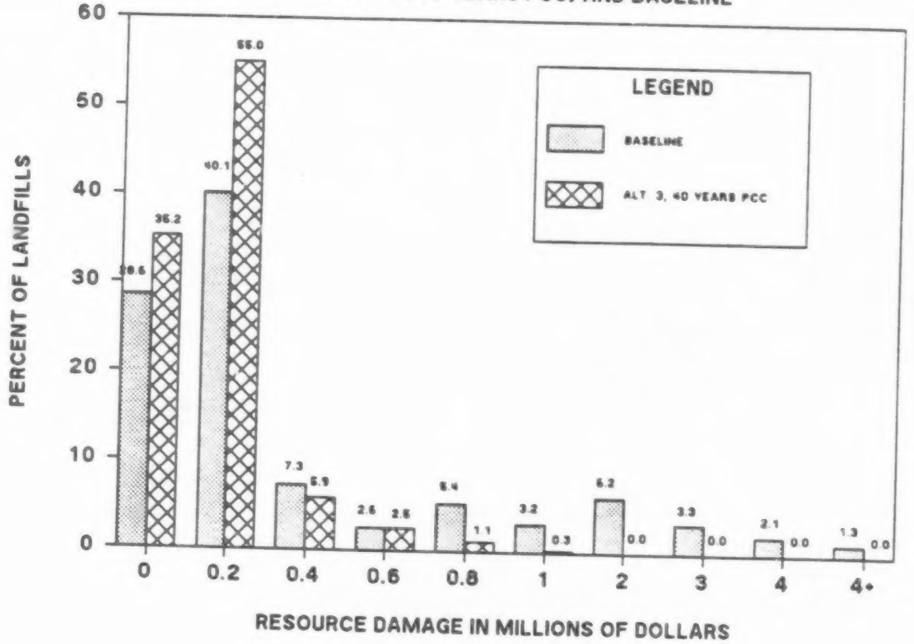
Under Alternative 3, the median baseline resource damage of $79,000 is reduced to $12,000 (40-year post-closure care period) or $28,000 (10-year post-closure care period). The total resource damage across 6,034 landfills ranges from $0.44 billion to $0.90 billion under Alternative 3, depending on the length of the post-closure care period.

Figure IX-35 shows the reduction in resource damage achieved for each size/setting/flow field combination. For landfills that meet the performance standard in the baseline (i.e., with vegetative covers and no liners), there is no reduction in replacement costs; however, these landfills almost always have very low resource damage in the baseline. For landfills that install some type of containment system to comply with Alternative 3, resource damage is reduced significantly. The effect of shortening the post-closure care period from 40 years to 10 years can be seen by comparing the two graphs displayed on this figure. The line of points in the second graph lies closer to the 45-degree line, indicating that the reduction in resource damage is smaller.

FIGURE IX-34

DISTRIBUTION OF RESOURCE DAMAGE

ALTERNATIVE 3 (40 YEARS PCC) AND BASELINE



ALTERNATIVE 3 (10 YEARS PCC) AND BASELINE

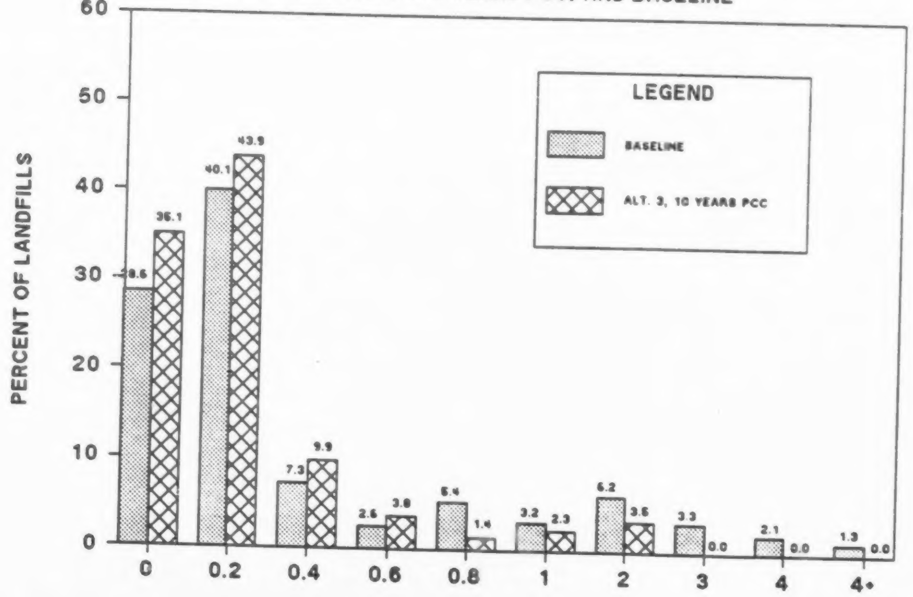
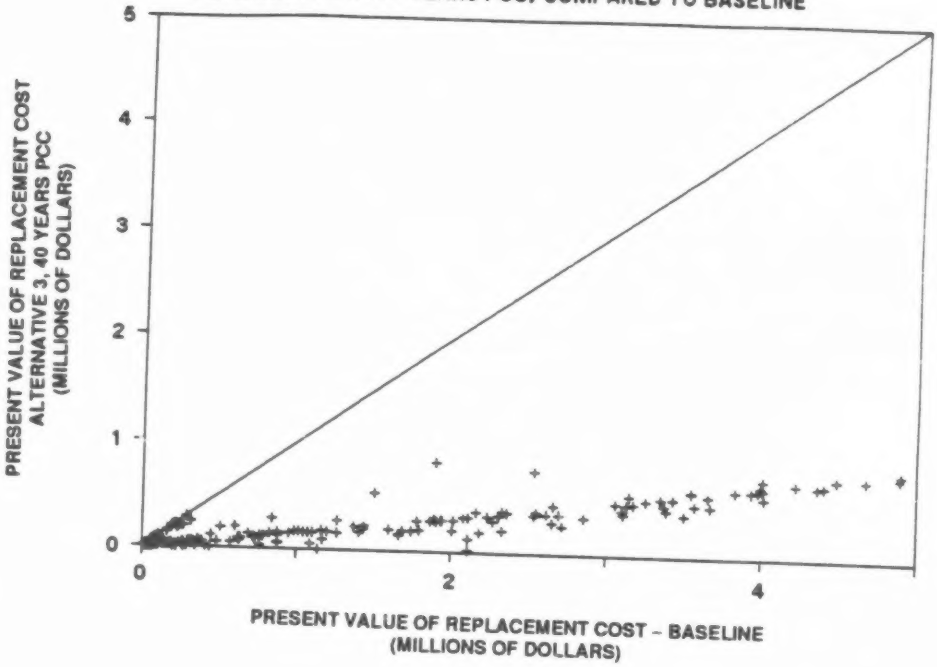
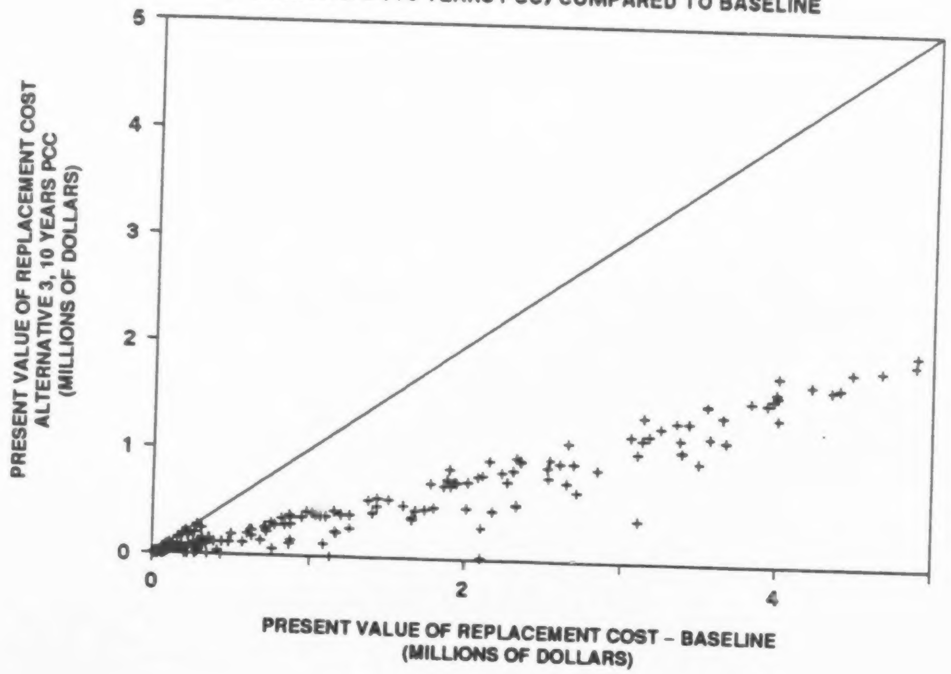


FIGURE IX-35
REDUCTION IN RESOURCE DAMAGE

ALTERNATIVE 3 (40 YEARS PCC) COMPARED TO BASELINE



ALTERNATIVE 3 (10 YEARS PCC) COMPARED TO BASELINE



Resource Damage Reduction – Alternative 4

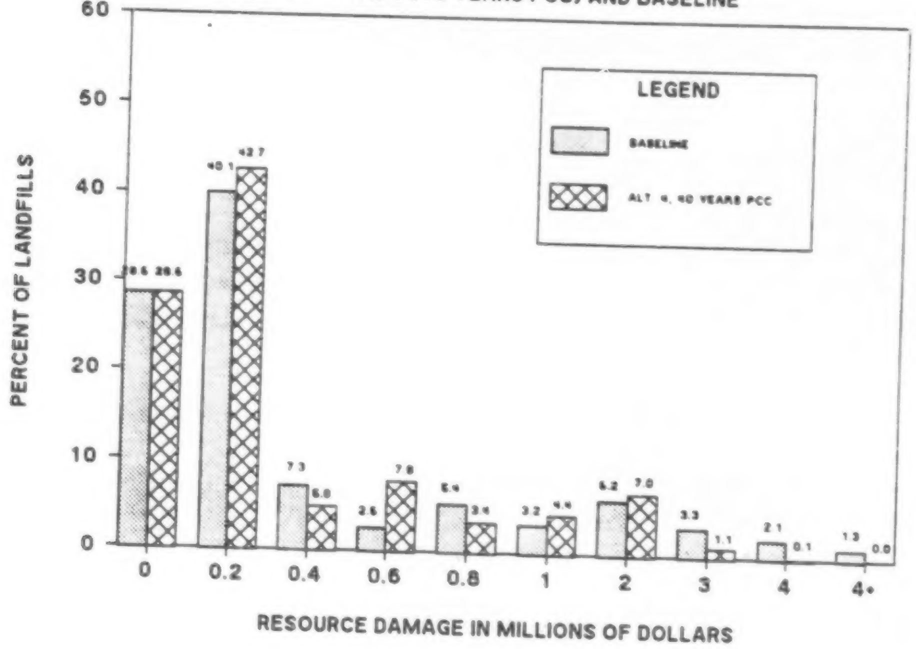
Alternative 4 requires owners or operators of landfills to perform corrective action when pollutant concentrations at the unit boundary exceed drinking water MCLs or, for pollutants without MCLs, the higher of a health-based level or an analytical detection limit. Corrective action can be triggered only during the active life and post-closure care period. The percent of landfills that trigger corrective action ranges from 33 to 35 percent under a 10-year and 40-year post-closure care period, respectively.

Figure IX-36 shows the distribution of resource damage under Alternative 4. As expected, the results are quite similar for the two post-closure care periods, as the only difference between them is the 2 percent difference in the percent of landfills that trigger corrective action. The figure indicates that this alternative reduces resource damage, especially in cases where the damage is most severe. Under Alternative 4, just over 1 percent of landfills have resource damages exceeding $2 million, compared to over 6.5 percent in the baseline.

Figure IX-37 shows the reduction in resource damage achieved for each size/setting/flow field combination. The points on the 45-degree line represent landfills that do not trigger corrective action. The figure shows that corrective action is triggered in a majority of scenarios with baseline resource damage in excess of $2 million, and in virtually all scenarios with baseline resource damage exceeding $3 million.

The median resource damage under Alternative 4 is $33,000 (for both 10- and 40-year post-closure care periods), compared with $79,000 in the baseline. The total resource damage across all 6,034 landfills ranges from $1.61 billion to $1.64 billion, depending on the duration of the ground-water monitoring period. Alternative 4 is the only regulatory option analyzed in which total resource damage is reduced by less than $1 billion. Corrective action alone is less effective in reducing resource damage than the use of more stringent containment system designs. Although corrective action can limit the size and duration of the plume, it does not delay the release of leachate from the landfill, and as a result there is less discounting of costs.

FIGURE IX-36
DISTRIBUTION OF RESOURCE DAMAGE


ALTERNATIVE 4 (40 YEARS PCC) AND BASELINE

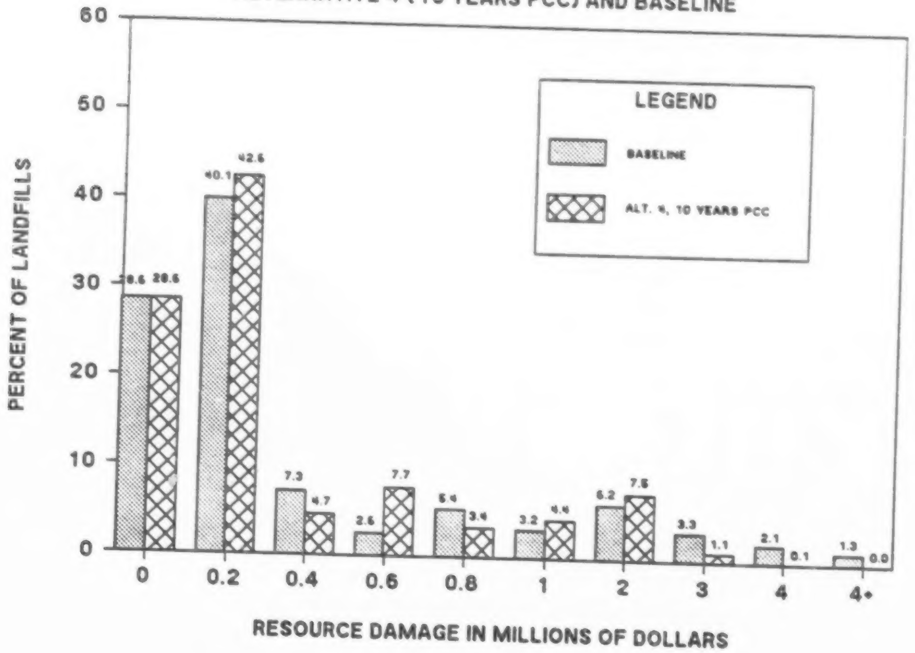
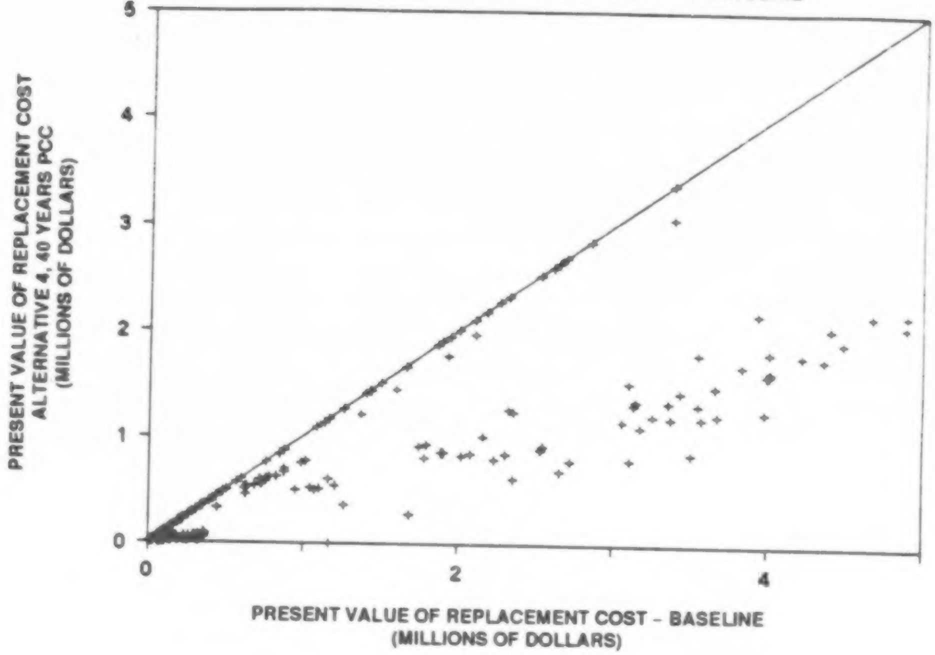
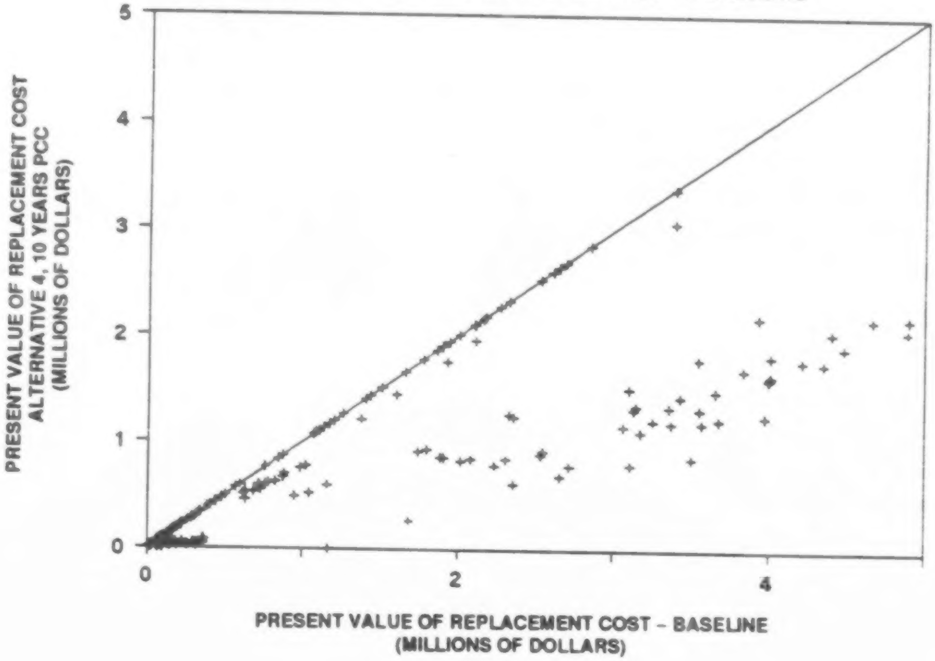

ALTERNATIVE 4 (10 YEARS PCC) AND BASELINE

FIGURE IX-37
REDUCTION IN RESOURCE DAMAGE

ALTERNATIVE 4 (40 YEARS PCC) COMPARED TO BASELINE



ALTERNATIVE 4 (10 YEARS PCC) COMPARED TO BASELINE



Resource Damage Reduction – Summary

All of the regulatory options reduce resource damage from baseline levels. For each option, the largest reductions occur at facilities that currently have downgradient wells within one mile as reported in the MSWLF survey. Table IX-3 shows the reduction in resource damage achieved for all of the regulatory options.

Table IX-3

Total Resource Damages for 6,034 New Facilities

(present value in billions of dollars)

Regulatory Scenario	Resource Damage	Damage Reduction
Baseline	$2.58	--
Final Rule	$0.84 - $1.24	$1.74 - $1.34
Alternative 1	1.21 - 1.29	1.37 - 1.29
Alternative 2	0.28 - 0.69	2.30 - 1.89
Alternative 3	0.44 - 0.90	2.14 - 1.68
Alternative 4	1.61 - 1.64	0.97 - 0.94

Alternatives 2 and 3 reduce resource damage most effectively. Alternative 2 imposes Subtitle C design requirements on all landfills. Under Alternative 3, which uses a performance standard based on time of travel and infiltration rates, over 70 percent of landfills must have synthetic liners and covers. The delay in leachate release achieved by these containment systems significantly reduces the present value of the water supply replacement costs.

The final rule and Alternative 1 are less effective. Under these regulatory options, more than half of the landfills can meet the performance standards with the unlined/vegetative cover design which does not delay releases.

Alternative 4, the statutory minimum, is the least effective in reducing resource damage. Under this alternative, no preventive design or operating measures are required. Although more than 30 percent of the landfills trigger corrective action under this alternative, corrective action is generally less effective than leachate containment in reducing the present value of the water supply replacement costs.

The length of the post-closure care period has an important impact on resource damage for landfills with synthetic liners and covers. For these designs, post-closure care delays the release of leachate, reducing the present value of replacement costs. The additional delay in releases achieved under longer post-closure care periods would also provide more time for pollutants to degrade prior to release, but we were unable to quantify the effects of this phenomenon. As a result, our analysis probably underestimates the significance of the length of the post-closure care period.

X. Comparison of Costs and Benefits

In this analysis, we have reported the costs of each option as the incremental cost compared with the analytical baseline of modeled, current costs. The benefit measures quantified in this analysis include reductions in human health risk and in resource damage; resource damage is calculated as the cost to replace contaminated water supplies for current or potential users. In Chapter VII we compared the costs of the different options and in Chapter IX we showed the relative effectiveness of each in reducing risk and resource damage. To conclude the analysis, we have compared the costs and benefits using consistent assumptions for each.

The comparisons have several limitations because of the assumptions made in the RIA. First, the two categories of benefits (avoided risk and resource damage) deal with ground-water-related effects: risk from drinking contaminated ground water and the avoided cost of new water supplies (resource damage). The cost estimates are not limited to measures that protect ground water, however; they include requirements that would reduce risks in other areas including surface water and subsurface gas. As a result, in comparing costs and benefits, we are ascribing to some options a higher cost than is incurred primarily for ground-water protection. At the same time, the ground-water protection costs account for a large percentage of incremental costs, so the comparisons are reasonable.

EPA believes that this rule will yield additional benefits that have not been quantified in this analysis. Additional benefits attributable to the final rule include improving public confidence, protecting the existence and bequest values of ground-water, and protecting property values. Therefore, this chapter provides only a partial picture of the benefits. Thus when assessing the effectiveness of each option, the additional benefits should be considered in conjunction with the quantified benefits.

The second limitation is that these comparisons address only new landfills. In order to present consistent measures of costs and benefits, we modified the cost results to compute costs for one generation of new landfills only. The cost model results presented in Chapter VII include costs for new and existing landfills in perpetuity; however, the risk model simulates risk and resource damage for one generation of new units that begin operation at the start of the simulation period and close after 20 years. We model releases for 100 years after the unit first accepts waste. The model then tracks any releases that occur for up to 200 years, so the overall model simulation period is 300 years for each unit.

For comparative purposes, then, we only used costs for new landfills with no replacement landfills. The costs assume that the units will open in the first year of the simulation period, complete with all necessary design requirements. The units operate for 20 years and are maintained for the appropriate post-closure care period. Corrective action costs for new units are added as appropriate. Even though we analyzed corrective action costs at existing units, these are not addressed in this chapter because we have no corresponding benefit measures at existing units.

The third limitation on the comparisons is a caveat against combining the benefits measures: It is not appropriate to combine risk and resource damage estimates to calculate "total" benefits. The risk estimates assume that people continue to drink water contaminated with the constituents of concern. The risk reduction is a function of lower levels of contaminants in the drinking water. Resource damage, on the other hand, assumes that as soon as contamination is detected, people stop drinking the water and switch to an alternative drinking water supply. It is not appropriate, therefore, to attribute any risk reduction benefits if resource damage estimates are used. These two measures are best seen as two different approaches for measuring benefits. Risk emphasizes human health effects to current users without regard for the time at which they are exposed. Resource damage measures economic benefits (avoided costs) for current and potential users and incorporates timing by discounting future benefits: if contamination is merely delayed, and not prevented, RD would show an improvement while risks would not.

Costs and Risk Reduction

In order to compare risk reduction with incremental costs, we must match both measures to capture the same effects over the same period of time. The risk model estimated the number of cases that would occur for each landfill over each year of the simulation. Extrapolating to the population, the model calculated the total number of cases per year for the baseline, final rule, and each regulatory option. To compare the number of cases with the costs, we summed the number of cases over the 300-year period. These are shown in Table X-1. Based on the analysis of risk model runs, Alternative 2 is the most effective at reducing cases, and Alternative 1 (the 1988 proposal) is the least effective, but the range is quite small. We discussed the reasons for the different effectiveness in Chapter IX.

Table X-1

Predicted Population Risk

(risk from one generation of 6,034 new landfills, modeled over 300 years)

	Cases per Year	Total Number of Cases	Cases Avoided
Baseline	0.0770	23.1	--
Final Rule	0.0245	7.3	15.8
Alternative 1 (Proposal)	0.0279	8.4	14.7
Alternative 2 (Subtitle C Design)	0.0142	4.3	18.8
Alternative 3 (Categorical Approach)	0.0179	5.4	17.7
Alternative 4 (Statutory Minimum)	0.0234	7.0	16.1

For costs, we summed all the costs that would be incurred for each new landfill (including corrective action for the Statutory Minimum) and calculated the present value of costs for all landfills and also for the subset of landfills that have ground-water wells within one mile downgradient as reported in the MSWLF survey. Both costs are presented in Table X-2. We expect that as more wells are drilled near landfills (or landfills are sited near existing wells), more units will have risk, so total risks and associated costs will be more similar to these estimates. As with other costs, we measured only the difference between costs with the regulation and the baseline for both scenarios. In addition, the costs are for one set of 6,034 landfills that are operated for 20 years and then closed.

For these cost estimates, we used the cost scenario reported in Chapter VII, which assumes 40 years of post-closure care and no adjustment for regionalization or waste diversion. This scenario is the closest to the risk model assumptions of 30-year post-closure and no change in waste landfilled because of regionalization or combustion. Because the costs allow for a longer post-closure care period, they are somewhat overstated, but this does not affect the relative ranking of the options.

Table X-2 presents the incremental number of cases avoided (cases under the baseline minus cases under each option) and the incremental present value cost for each of the regulatory options for all landfills and the subset with wells. The most expensive option is Alternative 2, with Alternative 4 the least expensive.

Table X-2

Costs and Cases Avoided

New Landfills Only; 300-Year Period

		Incremental Cost (present value in millions)	
	Cases Avoided	**All Landfills**	**Landfills with Wells**[1]
Final Rule	15.8	$14,839	$6,564
Alternative 1 (Proposal)	14.7	13,021	6,167
Alternative 2 (Subtitle C Design)	18.8	46,660	21,048
Alternative 3 (Categorical Approach)	17.7	18,589	8,911
Alternative 4 (Statutory Minimum)	16.1	11,751	5,671

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[1] These costs are computed only for the fraction of MSWLFs that reported drinking water wells within one mile of the units.

The incremental costs vary much more than the cases avoided; some of the alternatives, therefore, impose much greater costs for limited additional benefit. Between Alternative 3 and Alternative 2, for example, landfill owners would incur an additional $28,071 million to avoid 1.1 additional cases over 300 years.

Alternative 4 is only moderately effective at reducing risk but it is the least expensive option. One important result of this analysis is that Alternative 1 is dominated by Alternative 4 (i.e., Alternative 4 costs less and provides more risk reduction). Because Alternative 4 has no design standards, its cost is roughly 10 percent less than the cost of Alternative 1, which includes a risk-based performance standard and more expensive ground-water monitoring. The risk-based design goal makes Alternative 1 quite effective at reducing high risks, but less effective than Alternative 4 (which relies exclusively on corrective action) at lower risk levels. This result is driven by the modeling assumption that corrective action is effective at all sites. In reality, however, corrective action is not always possible at some sites with certain environmental characteristics.

Costs and Resource Damage Reduction

Again, we adjusted the model results to allow us to match consistent results over consistent time periods. The risk model calculates resource damage for individual units in present-value terms. The sums of the present value of resource damage as estimated by the model are shown in Table X-3. The baseline resource damage, which represents the maximum amount of damage incurred, is $2,580 million. The reduction in resource damage is calculated simply as the difference between resource damage with no controls in place (the baseline) and when regulatory controls are imposed.

Table X-3

Present Value of Resource Damage

One Set of New Landfills Only[1]

(dollars in millions)

	Resource Damage		**Net Reduction In Resource Damage**	
	10-Year Post-Closure Care Period[2]	**40-Year Post-Closure Care Period**	**10-Year Post-Closure Care Period**[2]	**40-Year Post-Closure Care Period**
Baseline	$2,580	$2,580	--	--
Final Rule	1,240	840	$1,340	$1,740
Alternative 1 (Proposal)	1,290	1,210	1,290	1,370
Alternative 2 (Subtitle C Design)	690	280	1,890	2,300
Alternative 3 (Categorical Approach)	900	440	1,680	2,140
Alternative 4 (Statutory Minimum)	1,640	1,610	940	970

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[1] Estimates cannot be compared to national costs as reported in Chapter VII. To compute net benefits, the costs must be adjusted as shown in Table X-4.

[2] Post-closure care period is 30 years for Alternative 1.

Table X-4 presents the net benefits of resource damage improvements under each of the options. The results are presented using both the 10- and 40-year post-closure care periods; no adjustments are made for regionalization or combustion. Alternative 2 provides the most benefit in terms of avoided resource damage, and Alternative 3 avoids the second most resource damage regardless of the length of the post-closure care period.

Table X-4

Net Benefit of Resource Damage Improvements

(present-value dollars in millions)

	10-Year Post-Closure Care Period[1]			40-Year Post-Closure Care Period		
	Reduction in Resource Damage	Incremental Cost	Net Benefit	Reduction in Resource Damage	Incremental Cost	Net Benefit
Final Rule	$1,340	$14,046	($12,706)	$1,740	$14,839	($13,099)
Alternative 1 (Proposal)	1,290	12,705	(11,415)	1,370	13,021	(11,651)
Alternative 2 (Subtitle C Design)	1,890	45,485	(43,595)	2,300	46,660	(44,360)
Alternative 3 (Categorical Approach)	1,680	17,796	(16,116)	2,140	18,589	(16,449)
Alternative 4 (Statutory Minimum)	940	11,116	(10,176)	970	11,751	(10,781)

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[1] Costs and resource damage estimates assumed 30-year post-closure for Alternative 1. Estimates *do not* reflect waste shifts due to regionalization or combustion.

To compare benefits with costs, we calculated the net benefit as the reduction in resource damage minus incremental cost. When this net benefit is less than 0, as it is for all the options, costs exceed benefits. The final rule falls in the middle of the range of net benefits: The statutory minimum and the proposal have higher net benefits (i.e., less negative). The rank ordering of the options is not affected by the post-closure care period.

As we observed for the cost and risk comparison, some of the additional expenditures for the more expensive options buy very little additional resource damage reduction. For example, a move from Alternative 3 to Alternative 2 (using 40-year post-closure estimates) would avoid $160 million in RD, but at a present-value additional cost of $28,100 million.